

**COSTS OF PROTECTED AREAS
IN THE UNITED STATES**

**A Dissertation Presented for the
Doctor of Science
Degree
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INTRODUCTION

Faced with rapid loss of species and habitats on which they depend, governments and NGOs worldwide invest billions of dollars every year to set aside lands for conservation.

A- BACKGROUND AND MOTIVATION

Protected areas, or land owned in fee by agencies and non-profits to further conservation goals, have traditionally been the go-to choice for conservation interests (National Research Council 1993). The UN Environment World Conservation Monitoring Centre (2017) estimates that, currently, close to 15% of all terrestrial and inland water areas are protected. This figure falls short of the Aichi Biodiversity Target of 17% in 2020, that was added to the Convention on Biological Diversity by its 196 signatories in 2010 (Buck & Hamilton 2011; CBD 2014). But as the Convention prepares to set new post-2020 targets (CBD 2020), this percentage is expected to keep increasing. Although acquiring a parcel of land is only one of the many strategies available to conservation planners (Bean, 2000), it has the strong advantage to provide the owner with the highest right of control over the use of the property, allowing conservation organizations to address threats from habitat destruction and pursue active management to advance biodiversity goals (Margules & Pressey 2000).

This does not come without a price and conservation organizations incur a wide variety of costs when acquiring and subsequently managing such areas (Naidoo et al. 2006). A common distinction is made between up-front, one-time costs involved in acquiring land (acquisition costs), and recurrent costs of managing a conservation program on a given area, over time (*management costs*). Acquisition and management costs of protected areas are the focus of this dissertation; however, we will often refer to and discuss other type of costs. Their definitions are provided in Box 1.

Optimal decision making and resource allocation tools have been developed to help conservation organizations manage limited budgets available to support protected areas and other programs (Cullen 2013). Provided with spatially explicit estimates of both conservation costs and ecological benefits, these tools can help decision makers identify areas offering the best return on investment (ROI). Studies have revealed that this type of approach could lead to large efficiency savings (Armsworth 2014). However, limited data regarding actual conservation costs has meant that most of those studies have relied on readily available proxies instead of actual cost data. For example, gross revenue or economic rent from agricultural lands is often used in lieu of acquisition costs (Naidoo & Iwamura 2007; Jantke et al. 2013). The potential problem with this approach is that the dynamics of conservation land transactions could be very different from those of agricultural lands, and there is a risk that such estimates

do not preserve the spatial pattern of variation in actual protected area acquisition costs. Efforts to study management costs also have suffered from the lack of available data, and many resort to taking a "snapshot" estimation approach, focusing only on what has been spent or what would ideally need to be spent to achieve particular goals, in a single year (Frazee et al. 2003; Naidoo & Iwamura 2007; Carwardine et al. 2008) ignoring temporal variability in protected area management costs.

Relying on poor cost estimates will make conservation planning recommendations less effective (Wenger et al. 2018). I will describe actual management and acquisition costs of U.S. protected areas and then show how better cost accounting changes the outcomes of optimization analyses.

BOX 1: EXAMPLES OF CONSERVATION COSTS

Acquisition costs

One-time, upfront costs of acquiring property rights to a parcel of land.

Management costs

Recurrent costs of managing a conservation program. Examples include maintenance costs, but also active management costs, such as planting trees, organizing prescribed burns etc. as well as costs linked to human activity on the site (e.g. creating a network of trails, a parking lot...).

Transaction costs

Costs associated with negotiating an economic exchange. These include the costs of searching for properties, negotiating with individual landholders and obtaining approval for title transfer.

Overhead costs

Ongoing costs of operating a conservation organization that cannot be traced to any particular protected area or management activity. Building maintenance costs, office supplies, work vehicles, etc.

Opportunity costs for the landowner

Costs of foregone opportunities, which are a measure of what could have been gained from the next-best use of a resource. In land conservation, the net present value of alternative land uses is foregone when the site enters protection. Within a well-functioning land market, the price of buying land (i.e. the acquisition cost) should reflect opportunity costs in willing buyer-willing seller transactions.

B- DISSERTATION OUTLINE

This dissertation comprises three different analyses.

Chapter I focuses on management costs associated with protected areas across the southern and central Appalachian region. Management costs are by definition recurrent costs, but most studies treat them as constant, averaged through time. We ask if this could lead to some misrepresentation of management costs and investigate what factors seem to better explain large variations in management costs we observe in time and space.

Chapter II and III are concerned with a different type of conservation cost, acquisition costs, as they are actually incurred by land trusts.

In Chapter II, we focus on areas protected across the coterminous US since 1980 by a large land trust, The Nature Conservancy (TNC), and by local, state and federal governments. First, we identify drivers of acquisition cost variation across the landscape. Then, we use this model to fill gaps in our dataset and produce estimates of protected area acquisition costs for the lower 48 states.

Finally, in Chapter III, we use the fact that TNC commissions an evaluation of the fair market value of every property they buy. With that information, we can identify "bargain sales", which are land tracts sold to the conservation organization below market value, a previously unstudied type of conservation support. We then examine spatial patterns in the willingness of landowners to share the cost of conservation in this way, and what it means for conservation planning and optimization.

CHAPTER I

MANAGEMENT COSTS

The time path of investments for the management of protected areas: a case study.

A- ABSTRACT

Conservation strategies often center on creating and managing protected areas. Funding for protected areas is limited and recurrent costs associated with managing these sites must be considered in planning their acquisition. However, most conservation planning studies either ignore these management costs or use snapshot estimates of them, even though costs of managing a protected area can vary greatly through time. Here, we surveyed management costs incurred over 15 years for 37 protected areas in the Central and Southern Appalachian Mountains, US, that were established by a large land trust, The Nature Conservancy. We found that management costs for protected areas varied greatly through both time and space. We explored what ecological and socio-economic characteristics explain this variation, using a model selection and averaging approach. We found that management costs increased with site area in a way characterized by economies of scale. They were also greater for sites presenting a more rugged terrain and surrounded by a denser combination of roads and urban areas. Prescribed burns were strong drivers of management costs in the years they occur, while acquisition costs were negatively correlated to future management investments. Land managers felt that those protected areas that received less management effort were in worse condition and tended to spend more on areas with greater estimated species richness. Better accounting of how management costs vary in in space and time can help conservation organizations allocate their limited resources effectively and to evaluate the likely long-term cost implications of expanding protected area networks.

B- INTRODUCTION

While buying land to create reserves is a common approach to conservation across the world, it is also costly. Because available funding for conservation is limited, there is a need to ensure that spending of conservation dollars is efficient. This provides the focus for the field of conservation planning optimization, in which many studies have shown that substantial gains in conservation can be achieved by incorporating costs into decision-making (Naidoo & Iwamura 2007; Laycock et al. 2009; Duke et al. 2013). To do so, researchers typically take a return on investment (ROI) approach to identify locations and actions that provide the greatest ecological benefits per dollar spent (Ando et al. 1998; Polasky et al. 2001; Murdoch et al. 2007). Solving such conservation planning problems accurately can only be done when good quality

data are available for both the ecological benefits and costs associated with conservation actions (Armsworth 2014; Kujala et al. 2018).

Cost data that can inform protected area planning are often difficult to acquire (Iacona et al. 2018). When cost data are available, they often focus solely on acquisition costs - the one-time, upfront cost of acquiring land for protection (Ando et al. 1998; Polasky et al. 2001). Relying only on acquisition cost data assumes that variation in those costs is representative of variation in all of the costs accrued over the lifetime of a protected parcel (Polasky et al. 2001; Bode et al. 2008; Carwardine et al. 2008). However, conservation organizations face a wide variety of costs in establishing and maintaining protected area networks (Naidoo et al. 2006) that are not necessarily driven by the same factors in space or in time.

Management (or stewardship) costs - the recurrent costs of managing a site over time - are another major cost component involved in protecting sites. Management costs can be substantial, sometimes even large enough to outweigh acquisition costs (Armsworth et al. 2011). In addition, there is little reason to assume that management costs spatially covary with acquisition costs. For example, investments in management of protected areas might primarily reflect conservation organization's goals or ecological processes on protected sites, which often fall outside the market economy, while acquisition costs are more likely to respond to market-driven factors, such as the value of alternative land uses (Armsworth et al. 2011). Consequently, using only acquisition costs as a proxy for the overall cost of protecting a given tract of land could lead to biased cost predictions and make protected area planning less effective. Nevertheless, management costs have not been included in spatial prioritization analyses as often as acquisition costs. Having access to better management cost data could also allow a richer set of decisions to be considered in spatial planning analyses. For example, it could enable analyses of how best to choose among intervention strategies for a given site, based on their cost effectiveness (Polasky et al. 2001; Carwardine et al. 2012; Chadès et al. 2014); how to allocate human and other resources involved in site management (Dumoulin et al. 2014); or how to trade-off acquiring new protected areas with improving management and restoration on those already protected (Kuempel et al. 2018; Adams et al. 2019). Securing sustained funding to cover management costs is also often more challenging for conservation organizations than is funding initial acquisition of a site (Clark 2007). Having a better understanding of management costs can therefore help conservation organizations improve financial planning for those sites they are responsible for protecting.

Management costs for protected areas are only occasionally reported by conservation organizations or in scientific studies. Moreover, when management cost estimates are available, they are often coarsely aggregated through space and/or implicitly considered constant through time (Balmford et al. 2003; Wilson

et al. 2007). Until recent attempts (Cook et al. 2017; Iacona et al. 2018), the lack of general guidelines for management cost reporting have also made it impossible to compare them across projects, due to disparate methods. For example, some conservation planning studies use socio economic proxies, such as GNP and other country-scale measures of economic development, to estimate management costs (Moore et al. 2004; Bruner et al. 2004). Others estimate ideal budgets that would be needed to achieve a given ecological goal (Frazee et al. 2003; Wilson et al. 2007; Lessmann et al. 2019) while some studies focus on actual yearly expenditure (Armsworth et al. 2011; Silva et al. 2019). Despite the recurrent nature of management costs, most studies so far are 'snapshot' studies. They look at management costs at one time, but across sites that may have been protected for differing amounts of time. For example, Wilson et al. (2007) stretch a snapshot estimate over the study period, implicitly assuming that management costs are constant through time. Armsworth et al. (2011) averaged their yearly estimates over several years while seeking to account for differing amount of time since acquisition as a covariate. The risk with all those approaches is that they gloss over most or all the potential temporal variation in costs of managing protected areas through time.

In this article, we describe management costs and identify what parameters drive them through both space and time. To do so, we surveyed management costs incurred every year, over fourteen years, for 37 protected areas in the Central and Southern Appalachian Mountains, US.

C- MATERIAL AND METHODS

a. Choice of case study

The Nature Conservancy (hereafter TNC) is the largest conservation non-profit in the US, where it owns and manages around 8,000 km² of land (The Nature Conservancy 2018) and has helped protect more through its partnership efforts. TNC's approach to conservation planning is a common approach among NGOs: defining a portfolio of ecoregional priorities where subsequent land acquisitions, among other conservation actions, are to be focused. These large-scale assessments are based on biodiversity, socio-economics and estimates of the threat of habitat conservation (Conservation Gateway - The Nature Conservancy 2018). We focused our study on one of those priority regions. The Central and Southern Appalachian region is considered a hot-spot of biodiversity in the US (Stein et al., 2000, chap. 6 - Chaplin et al.) and contains a large number of endemic species not currently well protected by existing protected areas (Jenkins et al. 2015). Forests in this region also supply ecosystem services to a large proportion of the population living on the East Coast of the US (Mockrin et al. 2014).

Choosing to focus on sites managed by only one organization ensured some

level of consistency in regard to management cost reporting, as well as to the decision making and governance processes that lead to the acquisition and management of these areas. Due to TNC's hierarchical organization into relatively autonomous state chapters, our sample still spans a variety of managerial practices. For this reason and because TNC's approach to land protection reflects a relatively common operating model found in other conservation land trusts, our results should be relevant to other protected areas in the US.

Our sample of protected lands is comprised of all the areas TNC acquired (fee-simple ownership) within the Central and Southern Appalachian Mountains since 2000, that were retained and managed by TNC until at least 2014 and for which forest preservation was one of the stated conservation goals. This left us with 37 protected areas, encompassing nine U.S. states and protecting two main types of forest communities: pine assemblages and hardwood oak communities (Figure 1). Despite the relatively small sample size, reserves in our sample varied widely in their characteristics. For example, the protected areas in our sample varied in size by 3 orders of magnitude although many were small (quartiles: 13 – 69 – 186 hectares), encompassed a variety of elevations, ecological habitats and differed in their accessibility to the public (Table 1). TNC spent a total of \$15.7 million to acquire these sites.

b. Data acquisition

i- Cost data

To estimate which management activities had taken place on the sample protected areas, when and at what cost, we surveyed the land managers in charge of those sites. Typically, one land manager was responsible for multiple protected areas within a region. We conducted detailed surveys, in person or over the phone, with 11 TNC land managers. Survey questions that we used in our interviews with land managers took a work-sheet form detailing different expenditures; relevant questions can be found in the Supplementary Information (thereafter S.I.). We asked, for each site, how much staff-time, whether there was any costs associated with supporting volunteers (when applicable), how many trips and what other expenses were directly attributable to the protected area, per year. Example of "other expenses" included outsourced projects (contracts), extra-fuel or gear cost for particular activities (e.g. prescribed burns), creation and maintenance of trails or parking areas, illegal dumping cleaning and fees, etc.

To focus on site-specific management, we requested that land managers omit overhead costs in their estimates, where these included administrative costs (such as office supplies and paying administrative staff) and infrastructure costs

(such as office renting and power consumption, as well as purchase and maintenance of general equipment and vehicles). We argue that those costs are organization dependent and, since we worked with TNC only, that they would be comparable across all sites in our sample.

Except for outsourced interventions, for which a contract remained, land managers often had to make educated guesses as to how much they had invested into the management of any given site in a given year. We revisit the potential bias this could introduce in the discussion. We also asked managers to estimate the average salary of those who worked on the site over the study time period and transformed estimated staff time into a monetary cost. In the same manner, we multiplied the estimated number of trips per year by the distance between site and TNC office in charge. All TNC vehicles used to visit protected areas within our sample were pick-up trucks. Using national transportation statistics for average fuel efficiency of pick-up trucks in the US (U.S. DOE and EPA n.d.; Bureau of transportation Statistics 2017) we assumed an average consumption of 15 mpg, which we multiplied by the average price of fuel for those states, that year (U.S. Energy Information Administration 2018). All costs were translated into 2014 dollars, using the Consumer Price Index (US Department of and Labor Bureau of Labor Statistics, CPI Calculator, 2006). While, occasionally, other approaches to handling inflation have been used in conservation cost studies (Davies et al. 2010) most have relied on standard inflation indices, perhaps because these make it easier for other researchers to replicate and update estimates obtained (Iacona et al. 2018).

Within the analysis, we focused on the management cost of a given protected area while including the area of the site as an independent variable in the statistical models. We chose to do this instead of using cost per hectare as our response variable, because dividing by area in this way may lead to spurious correlation, incorrect estimation of protected area size effects and inflated r^2 (Brett 2004; Armsworth 2014).

ii- Explanatory variables

We examined the effect of time on annual management costs in several ways. In addition to the number of years since a site was protected, we used the year of acquisition itself as a factor. We also incorporated years of management as random factor to capture a possible effect of the general economic context (e.g., recessionary conditions) on management spending. Finally, some of our predictors were time-varying factors, such as prescribed burns (happening or not that year), abundance of protected areas and easements nearby (see below), distance to managerial office in charge, and an indicator of the annual budget of each TNC state chapter involved. For the latter, we used the total amount spent by a given state chapter on land acquisitions and easements per year during the period of our study.

We included area size, which has been linked to management costs (Balmford et al. 2003; Frazee et al. 2003; Lessmann et al. 2019), acquisition costs, state and cluster when several sites were part of the same management unit. We obtained elevation at the centroid of each site from the NASA-SRTM 1 arc second dataset (NASA JPL 2013) and extracted the average rugosity (3x3 neighborhood) over the site area with BTM 3.0 ArcGIS Toolbox (Walbridge et al. 2018). Additionally, we used Google Maps' itinerary tool to measure the distance to the TNC office in charge, for each site.

Some of our chosen explanatory variables described the characteristics of the landscape surrounding a protected site. We defined buffer zones of 3 diameters (1, 5 and 10 km) around each site's boundaries, then we measured the proportion of agricultural land (NatureServe 2014) and of protected land (USGS Gap Analysis Project, 2018) within the buffer. We had access to establishment dates of protected areas and easements, thus this value varied through time. When no establishment date was available, we assumed that land was already protected at the time of the site's acquisition. Finally, we also calculated a "visitability" index for each site as the product between urban area density and total road length in the buffer (U.S. Census Bureau 2015). In the main text, we present results for models fitted on data aggregated over the 5 km buffer; we include the analyses for the 1 and 10 km buffers in the S.I. as a sensitivity test.

When surveying the land managers, we asked whether they considered the land was already in ideal condition or not at the time of acquisition. Ideal condition was defined as how they would want the site to be in 50 years' time. We also asked whether they considered that the site's condition improved, stayed the same, or worsened since acquisition. Finally, we asked them to identify the ecological stage of the forests on site (old growth, in transition or mixed). In addition, we calculated the effective mesh size within buffer (Jaeger 2000) before and after protection, using data on protected areas from the PAD-US dataset (USGS Gap Analysis Project, 2018), as well as vertebrate species richness on the sites, as estimated using modeled species distributions from USGS for 52 species (USGS Gap Analysis Project, 2018b).

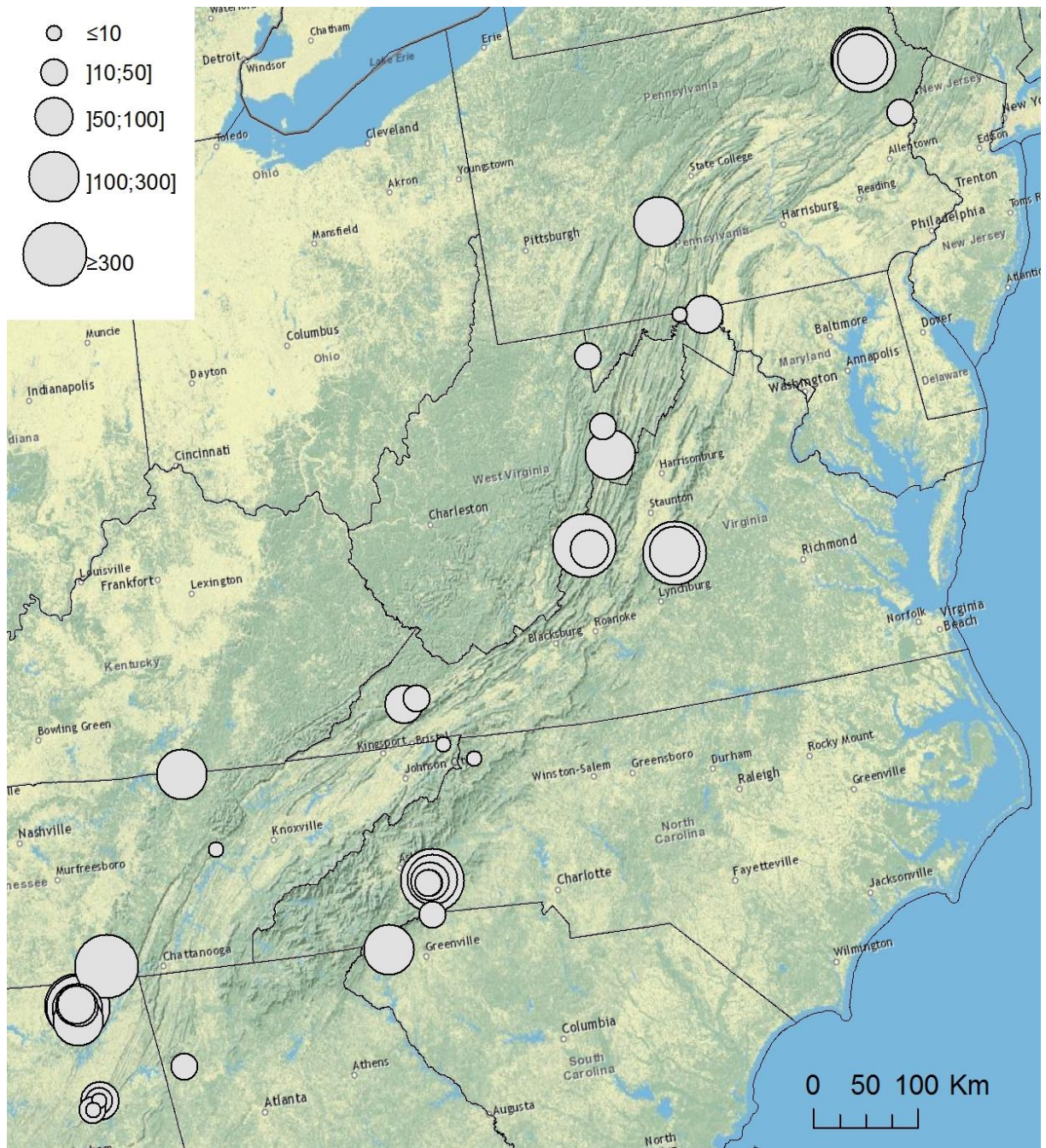


FIGURE 1: SAMPLE OF TNC PROTECTED AREAS

37 parcels in Central and Southern Appalachians - Size of circles represents relative size of protected areas, for ease of illustration only (scale is in hectares), all analyses used continuous area. States borders are represented by black lines.

TABLE 1: MODEL 1'S VARIABLE QUANTILES

Quantiles for continuous variables and area count per categorical variable used in Model 1

Variable	25%	Median	75%
Mngmt Cost (\$/site/year)	41	242	586
Area (km²)	0.13	0.69	1.86
Elevation (m)	262	370	577
Rugosity (index)	2.2	2.7	4.2
Acquisition Cost (\$)	31k	230k	630k
Distance to Off. (km)	86	157	240
Chapter Activity (new land protection per chapter \$/year)	0	550k	1,750k
Agri. Area (%)	4.7	10.1	18.9
Protected Area (%)	3.6	13.1	19.2
Visitability	126	254	357
Prescribed Burns	Yes = 15 sites/years		No = 33 sites
Ideal Condition (at acq.)	Yes = 14 sites		No = 23 sites
Forest Maturity	Old Growth = 15	Mixed = 4	Young = 18

iii- Model fitting, model selection and model validation

We fitted an initial linear model, complete with all the variables described above, to examine variation in management costs. We did not include any interaction terms in this initial model because we did not have a priori reasons to suggest that particular interactions might be relevant from among the many that are possible. We log-transformed the 3 cost variables (acquisition, yearly management and chapter activity) and site area to improve the model's fit. We tested all predictors for pair-wise collinearity and associated variance inflation factors.

Our data are nested in both space and time: we have multiple observations from the same parcels and we have observations across parcels that happened in the same years. In addition, TNC is structured into state chapters that operate with relative autonomy from one another. As a result, we expect that this structure might influence the spending pattern of management costs on protected areas within individual states. Therefore, we chose a model structure where sites (as management units), states, and years were included as random variables. We followed guidelines from Zuur et al. (2009) for model building and selection. We first fit the full model as described above. Then, while retaining this fixed model component, we selected the optimal structure of the random component of the model, based on AICc comparison. Retaining both the management unit and the state as random variables appeared to be optimal ($\Delta AICc > 2$ with the next best model, using REML estimators). We obtained the following model:

**Log(Costs) ~ Area + Elevation + Rugosity + log(Acquisition.Cost)
+ Distance.Office + Agricultural.Area + Protected.Area
+ Visitability + log(Chapter.Activity) + Prescribed.Burn
+ Forest.Maturity + Time.Since.Protection + Acquisition.Year
+ Habitat.Management + Ideal.Condition
+ (1|Management.Unit) + (1|State)**

MODEL 1

Next, we examined which of the various fixed effects should be retained. We generated all possible models given the set of explanatory variables, using R-package MuMIn (Barton 2018). We compared those using ML estimators and kept all models within $\Delta AICc < 2$ of the best model. This left us with 5 models, from which we then built an averaged model, using AICc weights (Table 2).

Finally, we checked the distributions of model residuals, compared them to the set of predictor variables and tested them for potential multicollinearity. The residuals conformed to expectations for a model of this type and did not show any sign of autocorrelation. We therefore proceeded with a non-spatial model structure for this study.

TABLE 2: REGRESSION TABLE

Best models (AIC Selection) and average model (using AIC weights) – The table presents parameter estimates for each model as well as AICc values, Likelihood-ratio based R^2 and AICc weights. Parameters' estimates are given with 95% confidence intervals. *Significance levels: * at 5%, ** at 1% and ***at 0.1% - Greyed out values signal a 95% confidence interval spanning 0.*

#	(Int)	Area	Rugosity	Acquisition Cost	Visitability	Prescr. Burns	Forest Maturity	Distance Office	Agricult. Area	Chapter Activity	Year Since Protection	R ²	ΔAICc	W
1	3.14±1.16	0.69±0.17	0.47±0.17	-0.16±0.05	0.0053±0.0014	2.63±0.59	mix= 2.32±1.04 old= -0.41±0.85					0.453	0.00	0.356
2	3.11±1.17	0.72±0.17	0.42±0.18	-0.14±0.05	0.0048±0.0015	2.66±0.59	mix= 1.48±1.30 old= -0.99±0.99		4.88±4.45			0.455	0.99	0.217
3	3.04±1.18	0.69±0.17	0.46±0.17	-0.16±0.05	0.0054±0.0015	2.55±0.60	mix= 2.30±1.05 old= -0.40±0.85				0.03±0.03	0.454	1.67	0.154
4	3.20±1.17	0.69±0.17	0.47±0.17	-0.16±0.05	0.0054±0.0014	2.63±0.59	mix= 2.34±1.04 old= -0.40±0.85			-0.01±0.02		0.453	1.91	0.137
5	2.92±1.25	0.69±0.17	0.50±0.18	-0.16±0.05	0.0052±0.0015	2.64±0.59	mix= 2.27±1.05 old= -0.47±0.86	0.001±0.002				0.453	1.92	0.136
Avg	3.10±1.18 **	0.69±0.17 ***	0.46±0.18 **	-0.15±0.05 **	0.0052±0.0015 ***	2.63±0.59 ***	mix= 2.13±1.16 * old= -0.54±0.92	0.000±0.001	1.05±2.88	0.001±0.007	0.004±0.02	0.453		

D- RESULTS

TNC spent \$959,000 to manage the sites over a 15-year period. The cost of managing a given area for a given year ranged from \$0 to \$168K, while the maximum spent in dollars per hectare in a given year was close to \$1.3k. However, management costs were in general relatively small; the median cost was around \$250 per year (or around \$5 per hectare per year), showing that the distribution is highly skewed toward small values. On average, the cost of staff time accounted for 73% of the overall expenditure, with only 5 sites where that metric was below 50%. Management costs were highly heterogeneous in both space and time. Average yearly expenditure per protected area ranged from \$6/year to \$3K/year across sites (or between \$0.3 and \$435 on average, per hectare per year), showing marked spatial variation of management costs across the Central and Southern Appalachian region. Median coefficient of variation (CV) in management costs per site since acquisition was 108%, evidencing a wide temporal variation in costs as well.

Each of the best models, given our chosen random component structure, explained more than 45% of the variation in observed management costs (Table 2). Most of the predictive capacity of these models stemmed from just three covariates: site area, acquisition cost, and prescribed burn occurrence. Management costs were larger for sites that were bigger and cost less to purchase initially. Management costs peaked whenever a prescribed burn happened. In addition, well-performing models also included rugosity, our visitability index and the categorical variable of forest maturity. These costs were larger for sites having more rugged terrain that were surrounded by a denser combination of roads and urban areas. However, other variables did not show significant associations with management costs. Most notably the only temporal variable that was consistently retained across well performing models was 'prescribed burns' from among the time-varying factors.

Local ecological benefit (perceived quality change of the site since time of acquisition, categorized as "better", "same" or "worse") was associated with management costs: sites where less money was spent were significantly more likely to be characterized as being in "worse" condition by land managers, and reciprocally (Figure 2a). Ecological benefit at the landscape level, when measured by difference in buffer's effective mesh size due to the acquisition of the site, was negatively correlated with management costs (Figure 2b). When measured by the number of vertebrate species whose range was at least partially protected by the site, ecological benefit at landscape level was positively correlated with management costs (Figure 2c).

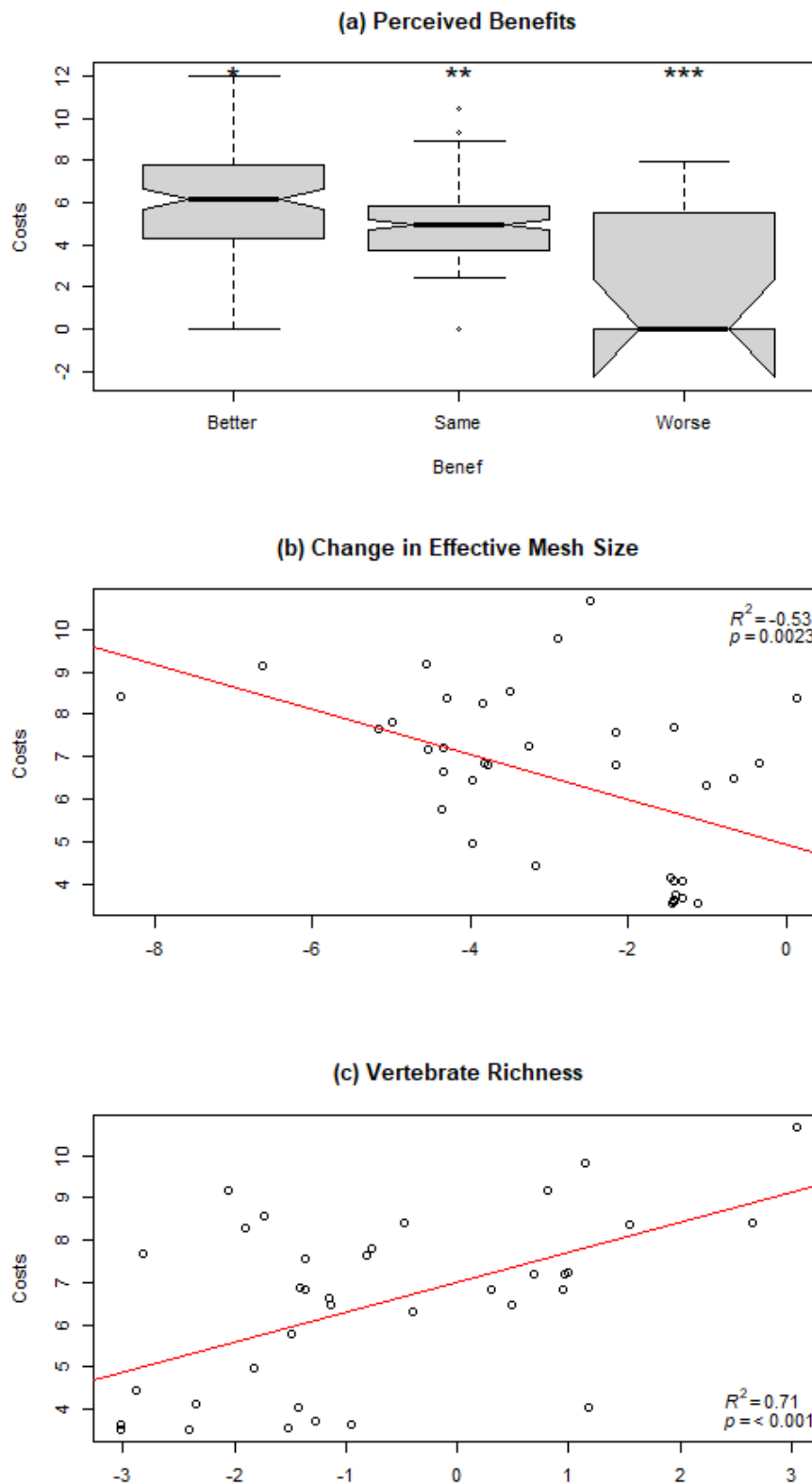


FIGURE 2: ECOLOGICAL BENEFITS

Ecological benefits are (a) ecological condition change as perceived by the land manager, (b) change in effective mesh size at the landscape scale due to site acquisition and (c) vertebrate richness protected by a given site.

E- DISCUSSION

Management costs are an important driver of the overall cost of protected area networks. However, management costs are often poorly documented and still little studied (Cook et al. 2017; Iacona et al. 2018). Better understanding of management costs is necessary to improve financial decisions pertaining to protected area establishment and maintenance. This could help scientists and conservation organizations produce more comprehensive spatial prioritization analyses, improve decision making between alternative management strategies on existing protected areas, optimize resource and personnel placement to achieve management activities more effectively, better navigate the trade-off between acquiring more land and improving management of already owned sites, and generally address the challenges of securing appropriate levels of sustained funding for a protected area. We examined how management costs of protected areas varied across a set of privately protected areas and how they varied through time.

When looked at the distribution of average yearly investment per site and found that the median was at \$10 per hectare per year. If funded on an endowment basis, this would require an initial investment of \$233 per hectare, assuming an annual rate of return of 4.5% (S.I.). When compared to the fair market value of the protected areas as detailed in real estate surveyors' estimates provided by TNC, we found that when costed on an endowment basis, management costs were on average less than 8% (ranging from 1% to 15%) of the fair market value of buying the sites. There was only one exception where management costs exceeded the site's fair market value (115% - the site is adjacent to a National Forest and the whole area is jointly managed by TNC and U.S. Forest Service, with frequent prescribed burns). In addition, variation through time of management costs also tends to be large (median CV 108%). Because of this, conservation organizations tasked with managing these protected areas need a sufficiently flexible budgeting model to be able to cope with this variation (Lennox et al. 2017).

Using a relatively straightforward set of predictor variables, we were able to explain 45% of the variation in management costs. Much of the predictive capacity of the models was due to just three variables: site area, occurrence of prescribed fires, and acquisition cost of the site. The coefficients associated with a site's area were always smaller than 1 and their 95% confidence intervals did not span 1. Because we are regressing log management cost against log area, a coefficient less than 1 signifies an economy of scale in management costs with the size of a protected area. I.e. increasing the size of a large protected area by one hectare would increase management costs by less than increasing the size of a small protected area by one hectare. Indeed, land managers frequently mentioned management activities that were performed independently of site area; for example, most state chapters required their staff to visit each site at least once a year, regardless of its size. At equivalent distance from the office, a larger site would then be comparatively cheaper to visit than a small one. Economies of scale in management costs have been found in several previous studies (Kim et al., 2014; Armsworth et al., 2011; Ausden, 2008; Balmford et al., 2003). Prescribed burns were a strong driver of management costs: they are very intensive both in term of total staff time spent at the site - before, during and after the fire- and incur large direct expenses, such as fuel and vehicles - TNC used fire trucks, flame throwers and helicopters routinely. As for sites acquired at higher prices, they received less management investments overall, which might be explained by the

fact that land trusts such as TNC will be prone to spend more when acquiring sites that are already at better condition and less in need of management investments. When checking the variables for collinearity, we noted that purchase price and the fact that the site was already at ideal conditions were somewhat correlated ($r=0.11$, $P\text{-value}=0.053$), supporting that hypothesis. Regardless our results suggest using acquisition costs as a proxy for conservation costs more broadly in conservation planning (Ando et al. 1998; Polasky et al. 2001) may yield inaccurate conclusions.

While supporting less of the overall predictive capacity, our set of well-performing models also retained other variables. Sites where rugosity was high were also more expensive to manage. Such sites tend to be harder to reach and any management activity that needs to take place in such places will be more time- and resource-intensive (for example, TNC maintained cabins on several sites for the purpose of accommodating necessary overnight stays by their staff). Also, our visitability index was consistently a significant predictor of management costs. Land trusts and other conservation NGOs such as TNC rely in large part on fundraising to support their operations. The land managers we met told us they spend time and effort organizing site visits for major donors, as well as signaling protected areas, hiking trails, and other facilities such as overlooks (themselves generating maintenance costs) and facilitating their access by the general public. Showcasing the organization's work is an important priority for land trusts, in an attempt to generate more awareness and create new fundraising opportunities for their projects (Clark 2007).

Surprisingly, we found that direct effects of time or time-varying factors were rarely retained in model selection with the exception of prescribed burns. Our results suggest management costs vary through time but are not simply a function of time elapsed since protection. Nor was Year retained as a random effect, despite our study encompassing the financial crisis of 2008 and ensuing recession. This may be because the non-profit sector in general, including conservation non-profits and TNC in particular, was impacted less by the recession than many other sectors (Larson et al. 2014; Friesenhahn 2016). Other variables also did not show significant associations with management costs. We expected that sites located far away from their managing office would in general be more costly to manage (Dumoulin et al., 2014) but this effect was not significantly discernable in our model. We were also expecting to see a stronger effect of density of protected land around the site, as it seemed that TNC's land managers were more likely to conduct costly management activities on larger clusters of sites (invasive removal, parking and trail construction and maintenance, replanting, etc.) but this trend was not supported by our data.

State, as a random component, slightly improved the model compared to accounting only for the management unit ($\Delta AIC=2.73$). This suggests that TNC's internal structure might somewhat affect how much is spent on management of different sites. Because site managers whom we interviewed each worked with a different state chapter, this could also reflect the existence of some individual level bias in managers' recall of past expenditures.

Looking at return on investments, we found a significant correlation between management costs and managers' perceptions of site condition (sites deemed in worse condition also received less management). This relationship, however, is difficult to interpret because of the subjective nature of managers' perceptions. It is not possible to rule out that managers might perceive a site as improving or getting worse based only on the quantity of management

dollars they have been allocating to it. On a broader scale, we found a significant negative correlation between management investment and the importance of a site for broader landscape connectivity. However, there is a trade-off between prioritizing species richness, which tends to be associated with targeting smaller sites, and minimizing fragmentation, which is better achieved with larger sites (Armsworth et al. 2018). We found a positive correlation (Figure 2c) between average yearly management expenditure and vertebrate species richness on the sites, as approximated using the USGS species distribution models. Together, those results might point to TNC land managers favoring species-rich sites when allocating management dollars.

Finally, with our study design, we made a number of important choices of which we wish to highlight three. First, we worked with only one organization, TNC. This ensured that we had access to consistent reporting of costs and that protected sites were managed along shared goals. The associated drawback is that our results are obviously tied to the particular business model of that organization and might differ for other conservation organizations as well as for public agencies. Therefore, it will be important to repeat similar designs in other contexts and settings. Second, we focused on management only, but conservation organizations will obviously face a wider variety of costs (overhead costs, acquisition costs, opportunity costs), most of which are so far relatively poorly understood, and it would be interesting to see more studies on the many costs of conservation, their patterns, and what drives them (Bruner et al. 2004; Naidoo et al. 2006). Third, our ecological benefits metrics could be improved by including field-based or remote-sensing measurements. Very few studies to date have been able to address the questions of investment efficiency and management strategy in the real world (van Wilgen et al. 2017).

Better understanding of management costs is necessary to improve financial decisions pertaining to protected area establishment and maintenance. For example, by quantifying the temporal and spatial variability of management costs, conservation organizations could determine how much budgeting flexibility they need and can evaluate different financial mechanisms to meet this requirement. Management cost data of the type we provide also enable estimation of the overall cost involved in operating a protected area. Fuller accounting of protected area costs is needed to inform discussions of how many and which areas conservation organizations should prioritize for protection and to inform fund-raising strategies that can ensure effective stewardship of these sites, once protected. Studies like this one can also assist in developing financial planning tools, such as simple endowment calculators (S.I.) - providing an initial estimate of the cost burden involved in taking on management of new sites - and help conservation organizations plan through time and on the longer term for the properties they manage.

CHAPTER II
THE PRICE OF BUYING LAND FOR CONSERVATION IN THE U.S.

What factors can explain the spatial variability of acquisition costs of land set aside for conservation in the U.S.?

A- ABSTRACT

Land acquisition is a crucial, but expensive part of conservation. Optimization studies have revealed that accurately accounting for the spatial heterogeneity of conservation costs could lead to large efficiency savings, but they have based this claim on methods of estimation prone to influencing the spatial pattern of variation in costs that they find. For example, because of a lack of data regarding actual acquisition costs faced by conservation organizations, a common approach in systematic conservation planning is to rely on more readily available proxies, such as agricultural land value. However, the lack of mechanistic understanding of what determines actual acquisition costs, as faced by conservation organizations, means that there is a risk that this proxy does not preserve their underlying spatial pattern of variation, returning cost-inefficient recommendations. With the goal of improving these predictions, we fit data from ~36,000 historical land acquisitions by public agencies using ecological and socio-economic covariates, to create the first nationwide map of acquisition costs and explain their spatial pattern across the continental U.S. While other land use-related values are useful predictors within our model, we show that they are not, by themselves, good approximations for acquisition costs. Using a more comprehensive combination of variables, our model was able to pick up twice the variation in acquisition costs as did agricultural land value. We found that larger parcels are less expensive on a per hectare basis while forested parcels, parcels overlapping IUCN listed vertebrates, or parcels located near urbanized areas tend to be more expensive to protect.

B- INTRODUCTION

Protected areas have long been a primary strategy for conservation, especially in terrestrial systems (Margules & Pressey 2000; Loomis 2002). In the U.S. alone, between 40 and 50 billions of dollars are invested in protection of natural resources every year (Parker et al. 2012; National Fish and Wildlife Foundation 2013), 22 billions of which are spent on land conservation (Lerner et al. 2007). However, those investment are below what would be needed to halt the erosion of biodiversity (McCarthy et al. 2012; Waldron et al. 2013). In this context of insufficient resources and pressing ecological needs, the demand for science to provide tools and insights for conservation planning and efficient decision making is higher than ever (Cullen 2013). Unfortunately, past choices about which places to protect have often been criticized. Many preserves exist because the land they protect was devoid of commercial value. Others were created primarily for recreational or other purposes and may also not be sited in the places most important to protect biodiversity (Pressey 1994; Juffe-Bignoli et al. 2014).

In recent decades, however, the U.S. has seen shifts in the roles different organizations play in land protection and in the strategies being used. Private land trusts have become prominent agents of land conservation (Albers & Ando

2003). Since 2010, 437 accredited land trusts have protected an additional 3 million hectares in the country (Land Trust Accreditation Commission 2019). In the face of continued biodiversity erosion and limited funding (Lerner et al. 2007; McCarthy et al. 2012), larger land trusts have increasingly adopted systematic approaches to identify parcels for protection, relying on work-flows and resources to organize planning efforts and optimization tools when appropriate (Amundsen 2011). Many of these methods aim to maximize the ecological return on investment (ROI) when selecting a set of areas to acquire (Moilanen et al. 2009a). Conservation ROI has been defined in various ways, but most definitions are based around the ratio of the ecological benefit of a conservation action divided by the economic cost of the action (Boyd et al. 2012). ROI approaches promise large efficiency gains as long as they can rely on reasonable estimates for both ecological benefits and economic costs (Cullen 2013).

Making relevant recommendations and optimizing conservation decisions require a good understanding of both achievable ecological benefits and associated costs. In the domain of land conservation, the latter is often poorly documented, and researchers rarely have access to actual land acquisition costs, as experienced by conservation organizations. Reserve design and resource allocation studies have to rely on proxies instead, which are often highly spatially aggregated (Sutton & Armsworth 2014), limiting the predictive power and accuracy of the results. A commonly used substitute for actual land prices for conservation has been agricultural land value (Margules et al. 1988). Most land acquisition for conservation happens away from urban areas, and so using agricultural land value may seem like a sensible approach, because we would expect acquisition costs to reflect the highest use value of the land, which tends to be for agricultural uses in more rural areas. However, land targeted for conservation is often ecologically different from typical farmland. It often includes steeper terrain and higher elevation areas (Sutton et al. 2016) that are less likely to have experienced recent habitat clearing. In addition, the dynamics associated with such conservation transactions can be quite different from those accompanying traditional agricultural land purchases. Motivations to buy and to sell between conservation organizations and existing private landowners could potentially be very different from those involved in conventional agricultural land sales (Armsworth 2014).

For this study, we used data on ~36,000 land transactions made to protect land. These data include land transactions made by the largest private land trust in the U.S., TNC, and land transactions made by local, state and federal governments across the U.S. Information on the latter were provided by the Trust for Public Land (TPL) based on TPL's Conservation Almanac (The Trust for Public Land 2019). Together the data account for 40 years of land purchase for conservation across the U.S. To our knowledge, this is the largest geographically explicit dataset of conservation land costs, to date. Despite the richness of the data, however, some areas are consistently seeing more land protection activity than

others (Table 3). To extend conservation practices and potentially reach ecological systems that have so far gone under-protected, we need to be able to predict conservation costs for locations that did not yet see much land protection activity. Here, we examine acquisition costs of these protected areas to identify potential patterns and drivers of their spatial variation across the continental U.S. and to build a model able to formulate cost predictions for previously under-protected areas. We also compare such cost data to agricultural land value, as well as urban land value, in order to assess how suitable those actually are when no actual conservation data are available.

TABLE 3: NUMBER OF LAND TRACTS PER STATE

AL	AR	AZ	CA	CO	CT	DE	FL	GA	IA	ID	IL	IN	KS	KY	LA
231	260	519	1548	960	152	267	6919	615	182	197	1747	1018	64	212	165
MA	MD	ME	MI	MN	MO	MS	MT	NC	ND	NE	NH	NJ	NM	NV	NY
1243	254	737	561	659	458	151	810	796	63	196	404	2098	399	165	1040
OH	OK	OR	PA	RI	SC	SD	TN	TX	UT	VA	VT	WA	WI	WV	WY
1526	179	1001	912	298	352	76	376	464	138	511	368	1532	2339	152	146

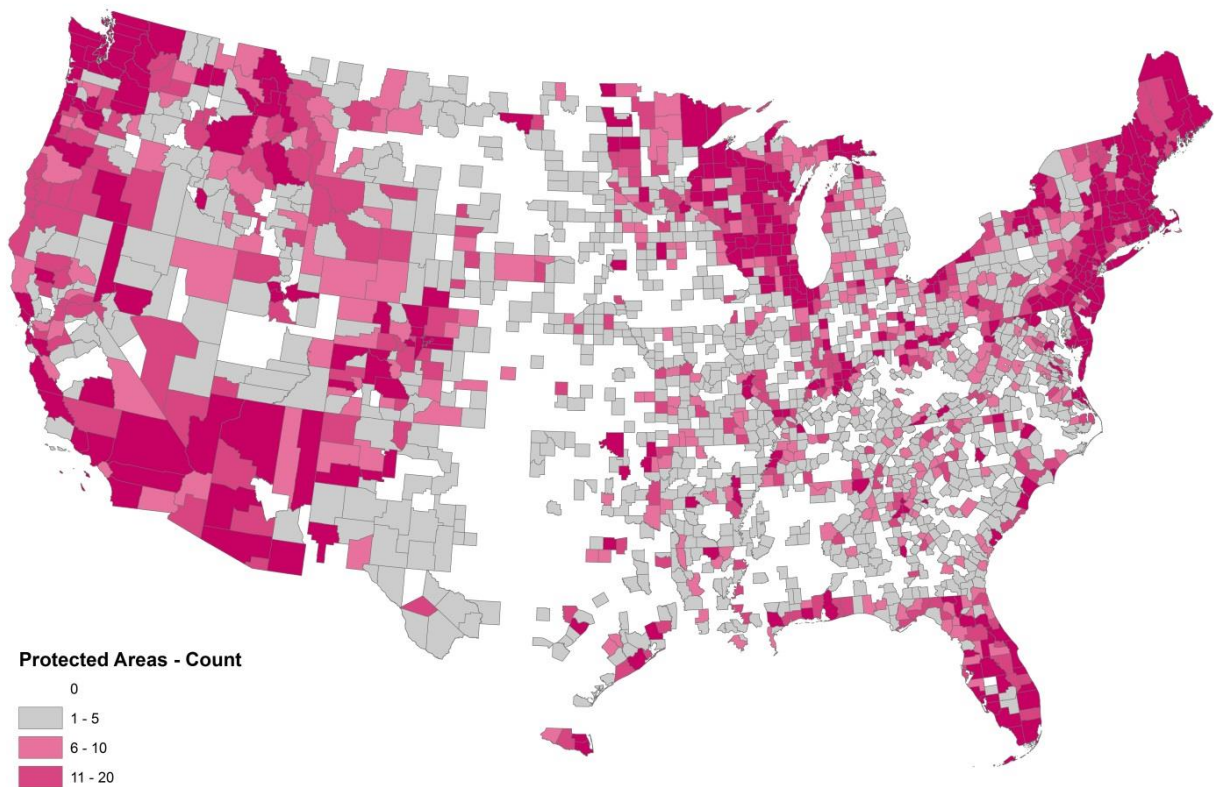


FIGURE 3: MAP OF COUNTY COVERAGE

Number of data acquisitions per county since 1980: records for the Great Plains region are scarce, with many counties containing fewer than 5 land deals (grey), while the Great Lakes region and both coasts are more densely represented (color scale).

C- MATERIAL AND METHODS

a. Response variable - Cost data at site level

The Nature Conservancy (2018) and The Trust for Public Land (The Conservation Almanac, 2019) databases hold records of land transaction as far back as 1917. However, we do not feel confident about the accuracy of deals that old. Older records were often incomplete and the socio-economic context has changed. Thus, we retained only transactions made in or after 1980. While this choice is admittedly subjective, sensitivity tests of different time periods revealed that our conclusions would not have been affected by the particular choice of start time (S.I.). After truncating the dataset this way, we were left with almost 36,000 entries across 1956 counties, which represent 63% of the 3108 counties making up the continental U.S. (32% if one is counting only those with more than 5 parcels). Those records account for more than 2.7 million of protected hectares, which represents almost 15% of all land conserved by land trusts in the U.S (Chang 2016). We corrected the costs for inflation using the Consumer Price Index of the U.S. Bureau of Labor Statistics (2019). We focus our analysis on predicting the average cost per hectare of purchasing land for protected areas within a county. Because we want to predict costs of buying land as faced by actual conservation organizations, we retained sites that were donated in our analysis. Around 16% of the land transactions are total donations (purchase price = \$0). This prompts the question of the prevalence of partial donations, or "bargain sales", when a landowner chooses to sell his land below market value for the purpose of protecting it, as issue we return to in the next chapter.

b. Independent variables

While recognizing other choices would also make sense, we chose to work at the county level for several reasons. First, based on conversations with practitioners, we believe that counties are a relevant spatial grain when one is deciding how to allocate conservation dollars across the country, for a nation-wide land trust. Final decisions over just which parcels should be acquired within counties are often left to local or regional land agents, but when one considers which parts of the country should be priorities for future investment as part of large scale budget planning, coarser spatial units are often used (e.g., forest blocks, ecoregions, or administrative units like counties). Second, counties are a relevant political unit in the U.S. in regards to regional and local land-use planning. Third, several of our chosen socio-economic variables are available only at county-level. Finally, and partly for these other reasons, it is also a scale at which many return on investment (ROI) based optimizations have previously been formulated, making it easier to compare our results with existing literature (Boyd et al. 2012; Withey et al. 2012). We used county boundaries from the U.S. Census Bureau (Tiger, 2015)

i- Ecological variables

The model we fit to explain variation in protected area acquisition costs included both ecological and socioeconomic variables, with the choice of variables based on a set of a priori hypotheses about factors that might explain cost variation. First, we considered the probability of habitat conversion for development in the county. Specifically, we summed the area of urban, crop and pasture land from the U.S. Forest Service's 2010 RPA assessment (Wear 2011) and considered it "developed" land. We used the 2030 projections included in the RPA assessment for developed land to obtain a measure of short term development threat, as a ratio of additional developed area to the current total developed area. We extracted number of hectares of protected areas of category 1 and 2 within the county from the PAD-US dataset (USGS Gap Analysis Project 2018). We divided all of the above variables by county size, treating them as densities. We also counted the number of protected areas per county in our dataset as well as their average size per county. We also obtained the mean elevation (NASA JPL 2013) for each county and calculated how many at risk species were present in the county, that is, species listed as CR (critically endangered), EN (endangered), or VU (vulnerable) by IUCN (2016). Finally, to account for potential geographical assemblage patterns, we also included EPA-1 ecoregional categories as factor (U.S. Environmental Protection Agency 2013) and tested for 2 additional levels of regional groupings that were not subsequently retained (S.I.)

ii- Socio-Economic variables

For the socioeconomic independent variables in the model, we first included measures of the value of alternative land uses, both agricultural land value (USDA-NASS 2012) and urban land value (Larson et al. 2019), because acquisition cost is likely to reflect the foregone value (opportunity cost) incurred when one protects land. For counties where these estimates were unavailable, we used the state average for the relevant variable instead. The acquisition price paid by a land trust also depends on the willingness of the landowner to possibly sell below market value (Chapter III), so we included median household income (U.S. Census Bureau 2017), percentage of adults with a bachelor's degree or more (U.S. Census Bureau 2017), unemployment rate, and population density (Friesenhahn 2016) as possible factors influencing these prices.

c. Model fitting

Even among the coastal states of the continental U.S., which are well represented in our dataset, there are counties with few to no records (Figure 4). Various methods could be used to estimate costs in these. For example, we could use a spatial smoothing or kriging approach. However, we think that getting to a better understanding of what factors drive costs is an important step toward

more efficient conservation practices, and smoothing methods would provide limited insight into this (Shmueli 2011). A better approach therefore is to explain variation in the cost data based on variables for which estimates are available in both space (for the missing counties) and time (for future predictions). This rationale drove us to choose a regression approach.

We started with a simple linear regression model. The analysis was conducted in R (R Core Team 2018), with packages MuMIn (Barton 2018), lmerTest (Kuznetsova et al. 2017), ape (Paradis & Schliep 2019) and DMwR (Torgo 2010). The average cost per hectare of buying land for conservation per county was log-transformed to reduce skewness. For the same reason as well as for consistency, we also log-transformed the average urban and agricultural land prices per hectare for each county. Our basic model structure was:

***log(Costs) ~ County.Area + Average.Parcel.Size + Ecoregion + IUCN.Listing
+ Elevation + log(Urban.Land.Value) + log(Agricultural.Land.Value)
+ Education + Median.Income + Unemployment.Rate
+ Development.Risk + Population.Density + Proportion.Developed
+ Proportion.Protected***

MODEL 2

The model was significantly improved by retaining large ecoregions, but when generating a proximity matrix with all pairwise distances between counties and applying a Moran's test to the residuals weighted by those distances, we found that remaining spatial auto-correlation in the error terms was still visible. We adjusted the error structure to account for spatial auto-correlation and re-fitted the model using generalized least squares with five different autocorrelation structures (S.I.) We retained a rational quadratic correlation structure as the best one for our dataset.

To make sure our model could produce reliable cost estimates for national level conservation planning, we tested its robustness in both time and space. We subjected our model to both an out of sample cross-validation routine (repeated k-fold with 100 repeats of 10-fold random sets - Kohavi, 1995) and an in-sample validation check by fitting the predicted values against the observed values across our whole dataset (Figure 5a). We also subdivided the dataset into 4 subsets, one per decade (since it covers 40 years of land purchase) and fitted those independently. In all cases, spatial and temporal, in and out of sample, the parameter estimates and predictions of Model 2 remained consistent (S.I.).

D- RESULTS

Parcel costs, costs per hectare, and parcel sizes per county were heavily skewed (Table 4). In general, protected areas were small, with more than 87% of them

smaller than 100 hectares and 26% below one hectare, as is commonly observed worldwide (Deguignet et al. 2014). Once averaged per county, the spatial variation across the U.S. is still large, with average price per hectare and average parcel size both varying by ~6 degrees of magnitude, with most of their distribution in the smaller part of the range and few large and/or expensive areas stretching distribution toward higher values.

Using Model 2, we generate a complete map, from coast to coast, of acquisition costs of land bought for conservation in the U.S. Figure 6 maps the predicted land acquisition costs from the model, including extrapolating to counties where we did not observe transactions. As would be expected, predicted costs of acquiring protected areas tend to be higher in coastal counties on the East and West Coasts, and around major conurbations in the interior US (Chicago, Atlanta, Phoenix, etc.). In contrast, acquisition costs appear lower in rural counties in the interior of the U.S., particularly in the Great Plains, where admittedly, more extrapolation is involved. The Model fit is highly significant (P -value < 0.0001) and is able to explain 34% of the overall variation in protected area costs that we observe.

Among our covariates, urban land value and agricultural land value are both significant predictors of protected area acquisitions costs, as would be expected (Table 5), the association with agricultural land value being particularly strong after controlling for the effect of other variables. The average parcel size within a county had a strong negative effect on the price of a hectare of land, which means that a tendency toward larger parcels will be associated with lower average price per hectare. Overall county elevation and number of IUCN listed species had a positive effect on acquisition costs, as did socio-economic factors such as education, population density and unemployment rate.

We had hypothesized that indicators of philanthropic giving to environmental causes (Fovargue et al. 2019) would be associated with decreased costs because actual acquisition costs are determined both by the value of alternative land uses and by any tendency for landowners to sell to conservation for below fair market value as a donation. Environmental philanthropy has been associated with higher household incomes (Mount 1996), higher levels of education (Greenspan et al. 2012) and higher employment rates as well as living in larger urban centers (Chen et al. 2011). It appears that here the patterns are different. Higher employment rates are still associated with lower purchase prices per hectare, in line with our hypotheses, but education levels and population density tend to correlate with increased land prices, while median income is not significantly covarying with purchase prices.

Ecoregions originally had a stronger effect on conservation land prices and we assumed that this was due to their picking up the spatially correlated pattern expected from our dataset. Once the model was refitted with the appropriate

spatial auto-correlation structure they did lose some of this correlation strength, but most of the 8 categories remained significant drivers of costs. Parcels in forested ecosystems cost the most per hectare to protect.

E- DISCUSSION

Ongoing losses of biodiversity and ecosystem services (Condition and Trends Working Group 2005; Pimm et al. 2014) and limited resources for conservation mean there is a pressing need to allocate what resources are available optimally (Waldron et al. 2013; Le Saout et al. 2013). This requires having a good understanding of how much conservation will cost in different places. However, conservation costs are however often poorly documented. We examined what parameters drive acquisition costs of protected areas and used that knowledge to predict protected area acquisition costs across the conterminous U.S.

Some of the associations we found with our models are to be expected. For example, larger parcels cost less on a per hectare basis. Economies of scale in acquisition costs with the size of the parcel have been found in previous studies conducted over much smaller extents (Kim et al. 2014; Cho et al. 2017) and it is interesting to see that this aspect of protected area acquisition costs still emerges very clearly when working at the scale of the whole of the U.S. We also unsurprisingly find positive associations of protected area acquisition costs with agricultural land value and urban land value. Agricultural land value has often been used as a direct estimate of protected area acquisition costs (Ando et al. 1998; Withey et al. 2012; Kroetz et al. 2014). We would however caution against using either of those in isolation: relying on either of these variables as a direct estimate of acquisition costs for protected areas would miss much of the relevant variation (Sutton et al., 2016 came to a similar conclusion over a smaller scale, regional comparison). To illustrate this point in Figure 5, we plotted simpler bivariate associations between actual average cost per hectare of acquiring land for conservation (y-axis) per county against average urban (graph b) or agricultural (graph c) hectare value per county. Note the difference of scale between x and y axes on each graph: urban land value and agricultural land value can each explain only approximately 13% and 15% (respectively) of the cost variation and greatly under-represent the magnitude of this variation. Armsworth et al. (2020) compared the consequences for protected area priorities of relying on agricultural land value versus actual protected area acquisition costs using an optimization approach focused on conserving terrestrial vertebrates. In that work, we found that the relative ranks of counties in terms of protection priority proved relatively robust to use of the proxy (correlation coefficients of $r = 0.75$ if one values all species and $r = 0.89$ if one focuses only on those evaluated as being vulnerable or worse by IUCN). However, the particular top priorities for protection that emerged in the optimal solution proved highly sensitive to the

representation of costs used, drawing as they do on the extreme values of the ROI distribution.

We have noted earlier several of the social-economic factors associated with philanthropic giving have a different effect on land acquisition costs than we initially hypothesized. This perhaps suggests monetary gifts for conservation are associated with certain measures of wealth (higher employment rates, median household income, population density, and education levels). Meanwhile land donations may be associated more with particular patterns of land ownership and be higher in rural areas where population density, education attainment, and household incomes may all be lower. In addition, since population density and education are associated with urban centers (Chen et al. 2011), there are two other possible mechanisms explaining why they drive prices higher. The first one is that land value around cities tends to increase steeply, and we see in Model 2 that urban land value has indeed a significant positive effect on conservation prices. Secondly, areas where there is a higher threat of impending development can be associated with a higher willingness to buy, pushing land trusts to accept less favorable pricings (Murdoch et al. 2007; Boyd et al. 2012). Yet, our short-term development threat indicator was not a significant driver of acquisition costs, so there does not seem to be a clear additional and distinct effect of threat itself amidst the general effect of urbanization in our model. However, willingness to pay more per hectare was positively associated with the number of species listed as endangered by the IUCN. This increasing effect on land cost is possibly aided by the fact that the presence of species of interest offers leverage to landowners for bargaining prices up (Lennox & Armsworth 2013).

In this study, we made choices and assumptions that should be kept in mind when one interprets our results. First, we conducted this analysis at the county level for reasons detailed earlier. But we also recognize that fine-grain information is lost when doing so. Notably, sub-county variation of acquisition costs could translate into potential additional low-cost opportunities for conservation (Sutton & Armsworth 2014). We should note that these would also be missed by using county averages of agricultural or urban land costs as has previously been done, which would potentially make these indicators perform even worse at such small scales. Sub-county variation would thus play an important role in translating larger scale plans into local measures (Pressey et al. 2013) and there is a need to harness that potential in conservation planning (Gotway & Young 2002; Holzkämper & Seppelt 2007). Second, we have little information regarding acquisition costs for several states in the central U.S. For example, we only have ~75 land transactions or less for Kansas, North and South Dakota – see Figure 4 and Table 3. These tend to be states where land protection approaches other than fee ownership are more prevalent, particularly term contract agreements made as part of the U.S. Farm Bill's Conservation Reserve Program (Farm Service Agency USDA, 2019; Jackson et al., 2020). As a consequence, it is likely that our model would produce less reliable estimates

for those states. Third, though our model explains roughly twice as much variation in acquisition costs as substituting agricultural costs did, it still leaves a non-negligible amount of variation unexplained and, in particular, performs poorly with respect to land donations (Figure 5a shows that land donations (in red) encompass the whole range of predicted values). A deeper investigation of when and how much of their land landowners are willing to donate when selling for conservation is needed (Chapter III).

Understanding and being able to predict the cost of land bought for conservation are necessary conditions for the development of useful and reliable optimization tools. Such tools are needed in the face of ever-increasing threats to biodiversity and the limited resources available to conservation organizations. With this work, we are providing a national map of protected area acquisition costs to empower national scale conservation planning exercises. The model we present also provides insight into some factors that consistently make some acquisitions more expensive than others: parcels that are smaller, are associated with higher alternative land use values, overlap with IUCN listed species, are located at higher elevations, or are surrounded by denser human populations with higher education levels and lower employment rates, will be more expensive to secure for conservation. In this light, the fact that parameter importance and estimates remained consistent through time was not necessarily surprising since those parameters do not tend to vary much through time. County level unemployment rates or education levels are relatively stable across a period of decades and their relation from county to county is mostly preserved.

TABLE 4: COST VARIABLES' QUANTILES

In dollar/ha at parcel level, averaged at county level and with the distribution of protected areas' sizes for comparison.

Quartiles	25%	50%	75%
Parcel-level Acquisition Price per Ha	1,923	8,649	35,522
County Average Acquisition Price per Ha	2,853	7,353	28,828
Average Parcel Size	22	50	128

TABLE 5: REGRESSION TABLE – MODEL 2

Parameters estimates for Model 2, with associated standard errors. *Significance levels: . at 10%, * at 5%, ** at 1% and ***at 0.1%*

	Value	Std.Error	P-value
(Intercept)	-5.29	1.47	***
County.Area	0	2 E-07	.
Average.Parcel.Size	-2.5 E-04	6.43 E-05	***
Ecoregion - Desert	-	-	-
Ecoregion - Mediterranean	0.73	0.54	
Ecoregion - Tropical Forest	1.73	0.92	.
Ecoregion - Northern Forests	1.57	0.39	***
Ecoregion - Northwestern Forested Mountains	0.70	0.34	*
Ecoregion - Marine West Coast Forest	1.40	0.61	*
Ecoregion - Eastern Temperate Forest	0.80	0.37	*
Ecoregion - Great Plains	-0.50	0.36	
IUCN.Listing	0.06	0.03	*
Elevation	4.45 E-04	2.07 E-04	*
Urban.Land.Value	0.23	0.09	*
Agricultural.Land.Value	0.93	0.12	***
Development.Threat	5 E-05	6.2 E-04	
Education	0.05	9.5 E-05	***
Population.Density	2.7 E-02	1.3 E-02	*
Median.Income	-3 E-06	6.8 E-06	
Unemployment.Rate	0.12	0.04	**
Proportion.Developed	0.50	0.32	
Proportion.Protected	-0.04	0.62	

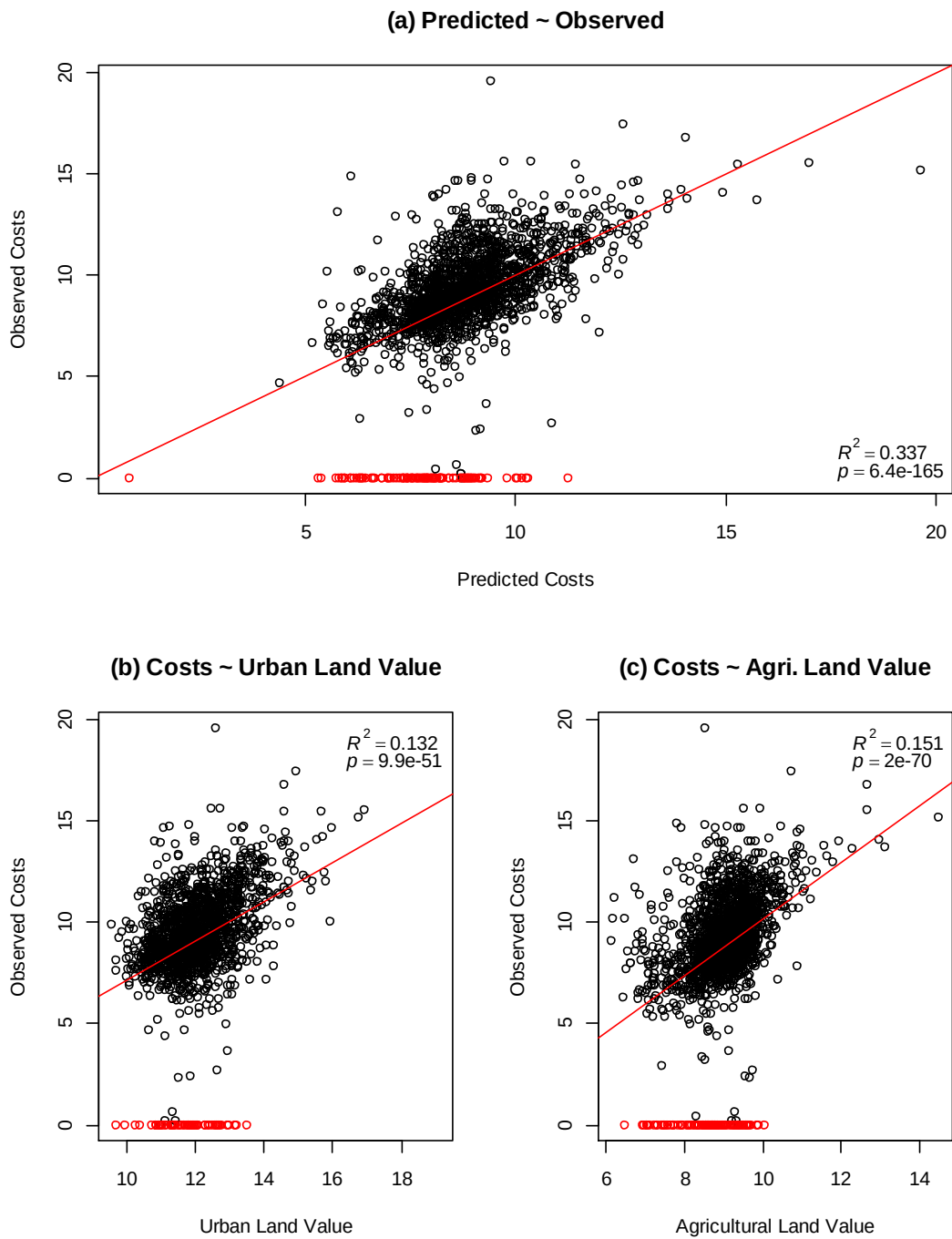


FIGURE 5: MODEL 2 FIT AND LIMITATIONS

Observed costs against costs predicted by Model 2, log-transformed and at county levels (a). Linear correlations between acquisition costs and urban land value (b) or between acquisition costs and agricultural land value (c). In all cases, red points along the horizontal axis represent counties where all acquisitions were fully donated.

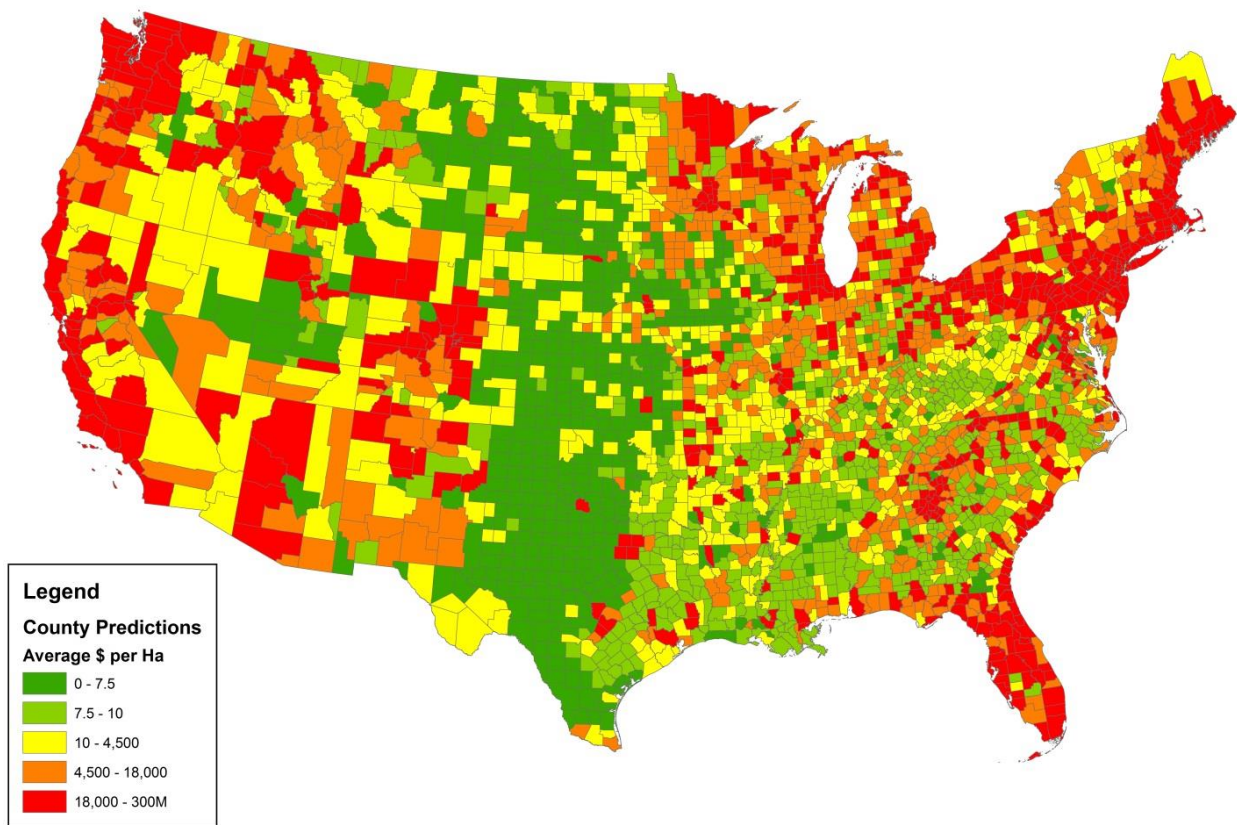


FIGURE 6: COMPLETE MAP OF AVERAGE ACQUISITION COSTS

Acquisition costs (in dollars per hectare) for all counties of the conterminous U.S. Values are real costs when there were >5 recorded purchases within a county or predicted costs from the Model 2 (whose parameter estimates are presented in Table 3) otherwise.

CHAPTER III
BARGAIN SALES IN THE U.S.

Predicting bargains sales occurrence and magnitude in land transactions for conservation in the U.S.

A- INTRODUCTION

We are facing an unprecedented challenge (Sutherland et al. 2009; Rands et al. 2010): biodiversity worldwide declines at an alarming rate (Butchart et al., 2010; Millenium Ecosystem Assessment, 2005; Global Diversity Outlook 3, 2010), and conservation organizations struggle to find the resources needed to face those threats (Balmford & Whitten 2003; Leverington et al. 2010). When one seeks to protect biodiversity by buying and setting aside land, just as important as identifying conservation priorities is to identify conservation opportunities (Knight et al. 2010). Integrating socio-economic values into spatial prioritization analyses, as opposed to accounting only for ecological value under a cost constraint, is the focus of the growing field of opportunity in conservation science (Noss et al. 2002; Knight & Cowling 2007; Cowling et al. 2010; Moon et al. 2014).

Spatial variation in support for conservation provides one measure of such conservation opportunities. A growing number of studies emphasize the importance of accounting for bottom-up sources of support for conservation in the design of strategies to protect biodiversity. These include studies on philanthropic giving (Larson et al. 2016; Fovargue et al. 2019), local ballot initiatives to protect land (Banzhaf et al. 2010; Kroetz et al. 2014) and volunteer effort devoted to managing sites (Asah & Blahna 2012; Armsworth et al. 2013). Each of these sources of support can be included in conservation planning as a form of cost-sharing or cost reduction for conservation. Understanding people's willingness to pay, sell, enroll, or act for conservation would enable conservation actors to identify areas of opportunity, where prospects of cost-sharing arrangements are more likely to materialize. In a context of very limited resources available to conservation initiatives, such bottom-up support for conservation cannot be overlooked. Fostering people's participation and involvement has also been recognized as an important factor for the success and long term sustainability of conservation actions (Selinske et al. 2017). The integration of human and social capital into the conservation planning process could lead to more rapid and cost-effective gains for conservation.

Here, we identify another form of this support. From a dataset of ~4,500 land transactions made by The Nature Conservancy (hereafter TNC) in the U.S. since 1980, we found that over 50% of land sold for conservation was sold below fair market value. This willingness of landowners across the country to sell their land below fair market value for conservation results in "bargain sales". While this is a common feature in land conservation, we know of no prior efforts to map and predict where bargain sales are likely to occur. In addition, reaching a better understanding of landowners' willingness to participate in conservation seems highly relevant in the U.S., where 61% of all land is privately held (Wiebe & Gollehon 2006).

Knowing how and why bargain sales happen could change which regions are identified as priorities for protection. The current need for efficiency has prompted the development of many optimization approaches and toolsets (Cullen, 2013; Sarkar et al., 2006) that could include conservation opportunities and opportunities for cost-sharing. Understanding bargain sales could also change what conservation actions are recommended. For example, conservation efforts could be directed towards building connections with local landowners in areas where bargain sales are likely to happen, in order to stimulate their occurrence and/or magnitude. It has also been proposed that bargain sales could explain part of the discrepancy sometimes observed between regions defined as priority areas and regions where conservation actually happens (Halpern et al. 2005).

Some insights about private landowners' willingness to offer bargain sales can perhaps be derived from writings on people's willingness to participate in conservation easement programs (Merenlender et al. 2004; Miller et al. 2011). At the same time, it should be recognized that entering into an easement is different for a private landowner from selling their property for below fair market value, and different motivations are likely involved.

In this study, we identify drivers of bargains sales' occurrence and size using a sequential model, fitted on the TNC dataset described in the previous chapter. For each land purchase made, TNC commissions independent experts to appraise the property, giving us access to the estimated fair market value of that tract of land at the time of purchase. By comparing this value to the price actually paid by TNC, we can deduce whether a bargain sale was offered and, when it was, how much of a discount was involved. We also examine the relationship between bargains sales and two other sources of social support for conservation: the number of conservation ballot measures proposed in a county (The Trust for Public Land 2020) and the average size of philanthropic gifts to TNC in that county (Fovargue et al. 2019).

B- MATERIAL AND METHODS

In the whole dataset, including records from both the Trust for Public Land and TNC (see "*Key Datasets*" section) ~16% of all land transactions were full donations, rising to 22% for TNC purchases. By comparing the acquisition price paid by TNC with the fair market value of the land at time of purchase, from appraisers' valuations of the parcel, we see that an additional 34% of the deals were acquired for less than their fair market value. This suggests that they were partially donated to the land trust by their previous owner. We hypothesize that land donations might depend somewhat on the same predictors as philanthropy or ecological involvement and volunteering. As a result, we decided to test variables that have been shown to predict those behaviors.

TABLE 6: COST VARIABLES' QUANTILES

In dollar/ha for the average acquisition costs and fair market values, the donated fraction is unitless.

Quartiles	25%	50%	75%
County Average Fair Market Value per Ha	2,419	4,993	11,204
County Average Acquisition Price per Ha	802	2,876	7,029
Donated Fraction (DF)	0	18%	58%

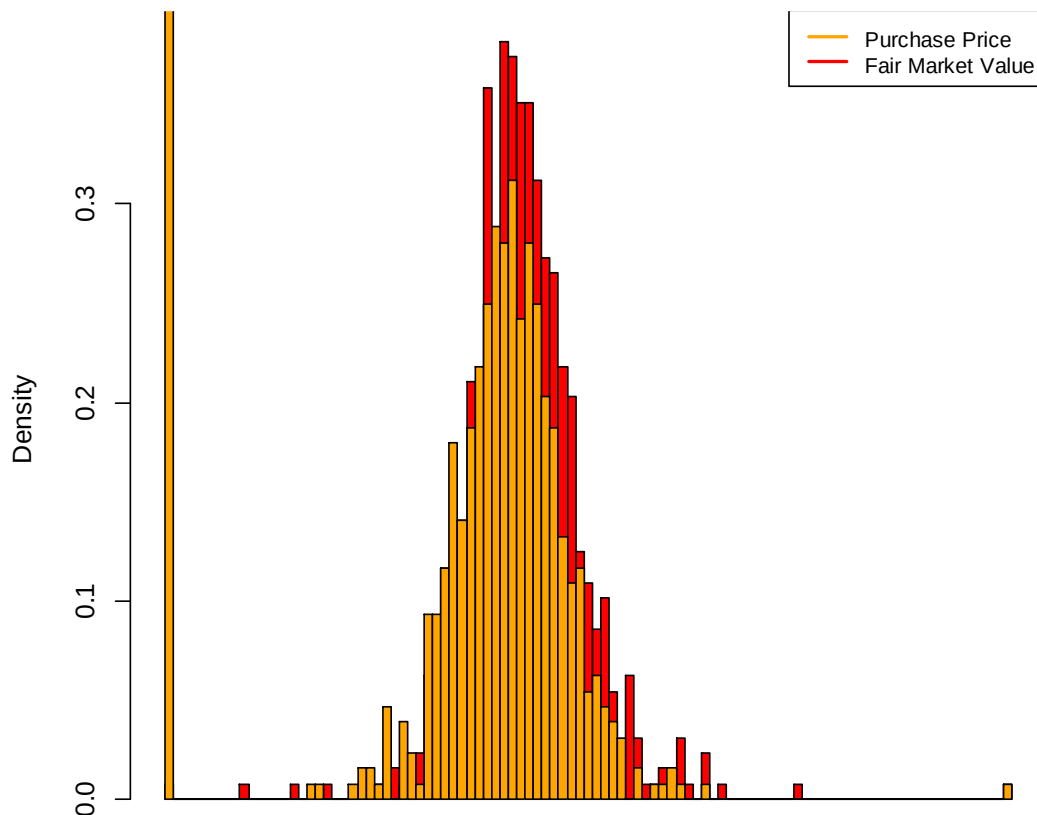


FIGURE 7: DISTRIBUTION OF ACQUISITION COSTS AND FMV

Distributions of purchase price distribution (orange and in front) and fair market value (red and behind). Note in the figure that bars are not stacked, rather one set of bars (purchase price) is in front of the other (fair market value) to make this overall size comparison more apparent.

a. Response variable – Donated Fraction

We calculate the donated fraction (**DF**) of a parcel of land acquired by TNC as follow:

$$DF = \frac{\text{Fair.Market.Value} - \text{Purchase.Price}}{\text{Fair.Market.Value}}$$

EQUATION 3

Typically, **DF** varies between 0 when TNC has paid the fair market value for this parcel, and 1 when the parcel has been entirely donated to TNC. For 10 parcels out of ~4,500, TNC paid more than the fair market value for a tract of land, resulting in a negative **DF**. These instances are very rare, and the percentages involved necessarily small, because as a nonprofit land trust, there are legal restrictions on when and by how much TNC is allowed to do this. We decided to treat those 10 data points as outliers and exclude them from further analysis.

b. Sequential Decision Model

Most studies of private landowners' willingness to participate in conservation use surveys to identify the factors behind this decision process. While this might be the best approach at a local scale, it would not be practical at the national scale, limiting its scope to inform large-scale conservation strategies. Here, we chose instead to look at patterns of bargain sales across the continental US by using a sequential decision model (Cragg 1971). First, we create a binomial model to predict whether a landowner will decide to offer a bargain sale. We fit a Probit regression, which uses an inverse normal link function, i.e. a cumulative normal distribution ("participation" Equation 2). Second, we select a generalized linear model with log-normal distribution to predict how much of a bargain will then be offered by the land owner ("magnitude" Equation 5).

$$\Phi^{-1}(Y_1) = X\beta_1 + \varepsilon_1 \quad \text{Participation term}$$

EQUATION 4

$$\text{Log}(Y_2 | Y_1 = 1) = X\beta_2 + \varepsilon_2 \text{Magnitude Term}$$

EQUATION 5

Where Y_1 is binary, $Y_1 = 1$ means the landowner offers a bargain sale (that is, **DF** > 0), $Y_1 = 0$ means the landowner does not. Y_2 is the magnitude of cost reduction offered conditioned on the occurrence of a bargain sale and β_i is the vector of coefficients to be estimated in model i . X is the set of socio-economic

and ecological explanatory variables (detailed below) against which each model is fitted. We kept the same set of predictors for both models, because we had no reason to assume that those two steps of the decision process would be guided by different drivers. This formulation implies a conditional independence assumption between Equation 4 and Equation 5, meaning that magnitude of the bargain sale is not influenced by participation.

c. Explanatory variables

We based our choice of variables on a set of a priori hypotheses we formulated about the mechanisms favoring bargain sales. Community attachment to the land has been shown to favour support for conservation (Campbell & Smith 2006; Farmer et al. 2011; Miller et al. 2011; Armsworth et al. 2013). However, this metric is difficult to estimate without relying on interviews and surveys. We decided to include the density of protected areas within the county, from the PAD-US dataset, category 1 and 2 (USGS Gap Analysis Project 2018). Given that the threat of development can influence decision-making and alter investment dynamics (Murdoch et al. 2007; Boyd et al. 2012), we include the same short term development threat indicator as we used in Chapter II, namely proportion of additionally developed area by 2030. We also included median income (U.S. Census Bureau 2017), education attainment (U.S. Census Bureau 2017), population density, and employment rate (Friesenhahn 2016) as these are commonly found to influence environmental preferences (Department for Communities and Local Government, 2010). Finally, economies of scale have been identified in most types of conservation costs and at most spatial grains (Kim et al., 2014; Armsworth et al., 2011; Ausden, 2008; Balmford et al., 2003, Chapter II) so we account for their potential existence here by using the average parcel size in the county as a predictor.

a. Covariation with other means of support

We wanted to compare the spatial pattern of bargain sales with other types of societal support for conservation. We obtained the number of conservation finance measures that have been placed on the ballot, at the county level, since 1988 (The Trust for Public Land 2020). We also used data on TNC's middle tier donors in 2009-2014 (Fovargue et al. 2019) as a measure of philanthropic giving by county. We fitted simple linear regressions between the donated fraction of land purchased for conservation in a county and each of those two additional metrics of environmental engagement (Figure 8).

C- RESULTS

Our dataset covers ~4500 deals across 638 counties (>20% of U.S. counties). TNC, on average, benefited from a bargain sale consistently in 65% of the counties for which we have data. The donated fraction averaged 52% over the whole dataset (and 76% at the parcel level). TNC alone spent a total of \$1.2 billion on the land purchases recorded within our dataset, which means that they saved an overall \$0.9 billion from bargain sales, or almost \$23 million per year.

Purchase cost per hectare – when positive – and fair market value per hectare both varied by ~6 degrees of magnitude and within a comparable range (Table 6). Their distributions are similar, albeit with a slight offset, but the main difference comes from the number of zero values in purchase prices, due to full land donations (Figure 7). As a result the donated fraction **DF** is both 0- and 1-inflated, which could cause potential overdispersion and create bias in parameter estimates and standard errors (Zuur et al. 2009). The sequential decision model (Cragg 1971) we use is adapted to that type of situation.

Looking at the coefficient table (Table 7) we can see that Average parcel size has a significant negative effect on participation but not on the subsequent magnitude of the bargain. We see something similar with education, but with a positive effect, and the reverse with unemployment, which has a significant negative effect on the size of the bargain offered, but not on the decision to offer one or not. Urban development is significantly negative for both aspects of bargain sales.

When comparing the pattern of bargain sales with other sources of support for conservation, we found that total or partial donations of land were not correlated with philanthropic donations or with the number of conservation ballot initiatives in a county (Figure 8).

D- DISCUSSION

Conserving land is expensive and resources available to practitioners are too limited (Waldron et al. 2013; Le Saout et al. 2013). In this context, planning around and even being able to encourage cost-sharing opportunities are necessary. Despite having the potential to save millions of dollars per year and significantly boost land trust budgets, bargain sales have so far been mostly ignored during the prioritization and budgeting process of these land trusts. Understanding bargain sales better could enable conservation agencies to successfully identify both where they are likely to happen and what tends to encourage their occurrence.

Landowners' willingness to offer bargain sales at all (participation term) was greater for smaller parcels in less developed areas where the average education

level was higher. Once conditioning to only look within those counties where bargain sales are offered, we find that the fraction of the purchase price being donated (magnitude term) tended to be lower in less developed counties with lower unemployment levels. Thus, while the signal from development is consistent (more and larger bargain sales in less developed areas), the role of other factors varies across the analyses.

At least some of the metrics traditionally associated with environmental engagement have the opposite effect from what we were expecting (Table 7), hinting at the fact that land prices and, indirectly, landowners' predispositions to engage in bargain sales (that is, selling their land below FMV) might not be driven by the same factors as those driving fundraising or volunteering for conservation (Ryan et al. 2003). Here we see that education is positively correlated with the likelihood that a bargain sale be offered but not with its magnitude. Proportion of developed area is negatively correlated to both bargain sale occurrence and size. Those characteristics are usually associated with urban centers, which are conducive of philanthropy and engagement (Chen et al. 2011) but are also associated with a steep rise of land value that might explain why land donation does not follow other markers of pro-environmental behaviors. Similarly, we did not expect income to not be a significant signal in bargain sales, as one would assume that counties struggling economically would not have as many landowners inclined to donate their land.

It is also important to note that donations (partial or total) could translate into opportunistic acquisitions by conservation organizations, as opposed to acquisitions chosen through prioritization analysis or similar strategic effort. Those could bias our estimates of conservation costs in both chapters somewhat toward land offered for donation that is potentially not of great conservation interest. The reverse might also be true: the dataset we have obviously contains only records of the opportunities that were taken by TNC and some lands may have been offered as full or parcel donation that TNC chose not to protect. Either way, our datasets in the two chapters still reflect recent conservation practice.

Meeting conservation targets will require being able to harness bottom up initiatives. This study showed that total or partial donation of the land was neither clearly correlated with philanthropic donations nor with the number of conservation ballot initiatives in a county (Figure 8). This suggests conservation organizations should use different approaches to unlock support for conservation in different places. Drawing from such results, we could inform more efficient conservation strategies, such as directing efforts towards building connections with local landowners in areas where bargain sales are likely to occur. Understanding how and why bargain sales happen could help correct conservation practices and/or change the practical definition of priority regions, by including conservation opportunities in their definition.

TABLE 7: REGRESSION TABLE – MODEL 3

Parameters estimates for each part of Model 3, with associated standard errors.
*Significance levels: * at 5% and ** at 1%*

	Participation Model			Consumption Model		
	Estimate	Std. Error	P-Value	Estimate	Std. Error	P-Value
(Intercept)	-7.98E-03	4.19E-01		-5.94E-01	4.33E-01	
Average.Parcel.Size	-1.78E-04	8.72E-05	*	-8.04E-05	9.58E-05	
Development.Threat	6.74E-04	3.27E-03		-1.37E-04	2.72E-04	
Proportion.Protected	-1.58E-01	4.25E-01		1.73E-02	3.81E-01	
Proportion.Developed	-7.16E-01	2.49E-01	**	-5.42E-01	2.63E-01	*
Land.Value	-8.85E-08	4.00E-07		-7.21E-08	4.18E-07	
Education	1.73E-02	8.28E-03	*	1.06E-03	8.04E-03	
Median.Income	3.66E-06	6.07E-06		3.91E-06	5.82E-06	
Unemployment.Rate	1.43E-02	3.70E-02		-8.96E-02	4.07E-02	*

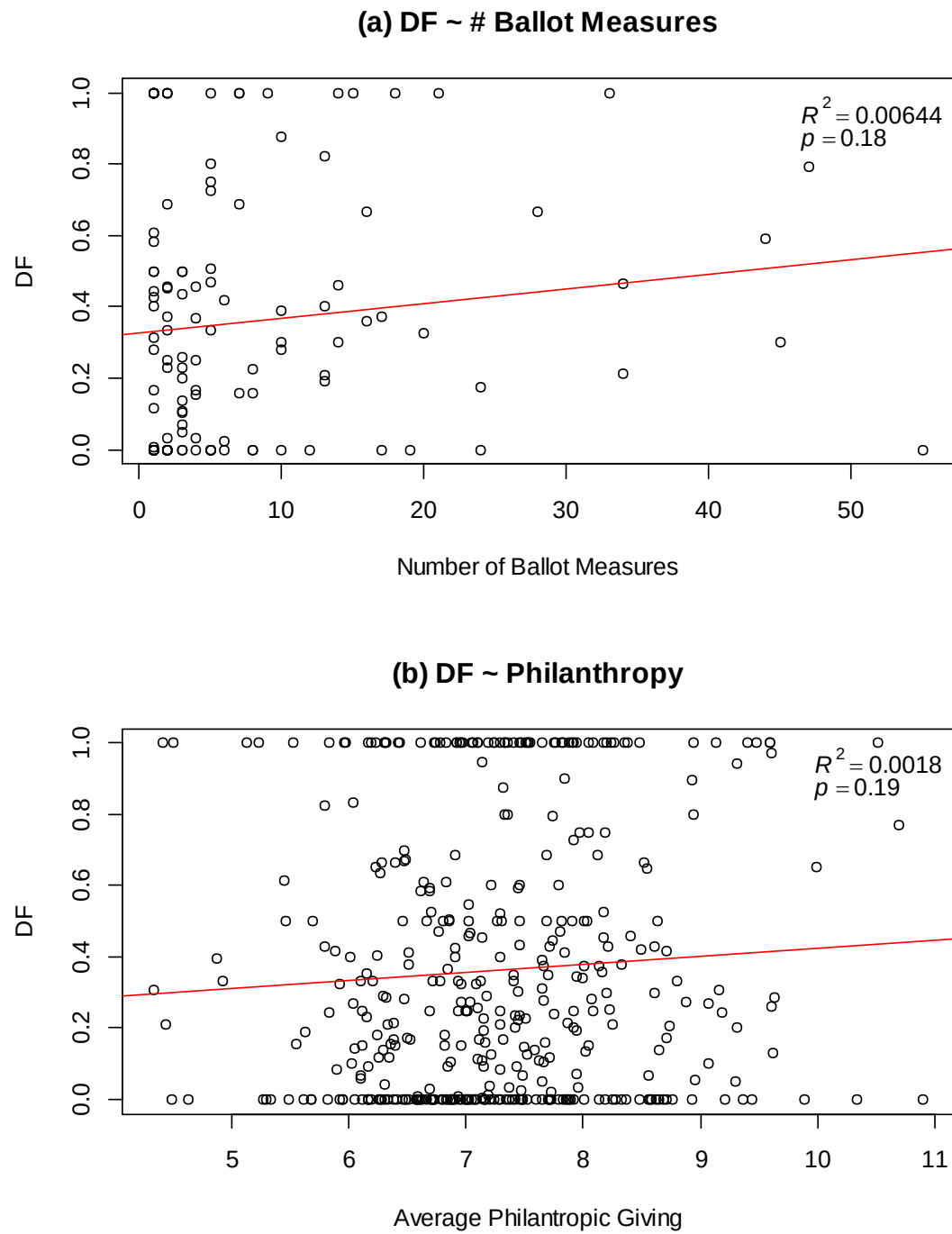


FIGURE 8: COST-SHARING OPPORTUNITIES

Bargain sales do not seem to correlate with the number of ballot measures proposed in a county (a) or with philanthropic giving

CONCLUSION

In an attempt to respond to the ongoing global biodiversity loss (Barnosky et al. 2011), efforts to protect land worldwide have intensified. The surface of protected land has almost doubled in the last 30 years, to currently account for ~15% of terrestrial areas (UN Environment World Conservation Monitoring Centre 2017). While protected areas have proven to be an effective conservation tool (Gray et al. 2016), this effectiveness depends of how sensibly the location for their establishment is selected (Watson et al. 2016) as well as how well and consistently managed they are (Jones et al. 2018). Acquisition and management of protected areas are costly, while resources available to support conservation are both limited (McCarthy et al. 2012) and, paradoxically, declining (Watson et al. 2014). In this context, spatial optimization approaches have become a central tool for conservation (Moilanen et al. 2009b; Groves & Game 2015), allowing practitioners to maximize ecological returns per dollar invested (Naidoo et al. 2006). However, this potential for efficiency gain is fundamentally limited by our understanding of both ecological benefits and costs associated with protected areas (Armsworth 2014; Kujala et al. 2018).

In this dissertation, we investigated what drives two types of conservation costs: management costs in Chapter I and acquisition costs in Chapters II and III. We further investigated societal opportunities for cost sharing and bottom-up contribution to conservation, such as bargain sales, and how they affect acquisition costs in Chapter III. Finally, we examined the consequences of this new understanding of various costs and their spatial patterns across the continental U.S. for conservation practices, notably how this could alter current strategies for spatial prioritization of land to protect.

A- MANAGEMENT AND ACQUISITION OF PROTECTED AREAS

Cost data pertaining to protecting land are often difficult to acquire, and even more so when it comes to management costs (Iacona et al. 2018). As a result, costs are not always considered, and when they are, acquisition costs are often the only focus. This makes the implicit assumption that acquisition costs correlate with all the other costs associated with the continued protection of the parcel of land they secured.

In Chapters I and II we showed that management costs do not follow the same spatial patterns as acquisition costs. Instead they each respond to different types of factors. Management costs are mostly driven by ecological properties of the protected areas and somewhat reflect land trusts' priorities. Acquisition costs, on the other hand, covary with two categories of factors: market-driven ones, such as the value of alternative land uses, and societal ones reflected in the

willingness of landowners to sometime sell their land to conservation land trusts for a bargain.

Chapter I's result were based on data collected at the site's level and through personal interviews of the land managers in charge of those areas. This methodology obviously limited the geographical scope of our study. There is however no reason to suspect that the discrepancy outlined above would not remain across broader spatial ranges, highlighting the need for larger scale estimates of management costs. We could then consider management costs at the country level, alongside acquisition costs, so as to gain a more complete picture of what spatial prioritization and optimal financial planning could look like.

B- ACQUISITION COSTS ARE NOT ALTERNATIVE-USE LAND VALUES

Due to the lack of availability of good quality data for acquisition costs of conservation land, most studies have used agricultural land value in lieu of these (Murdoch et al. 2007; Strange et al. 2007; Withey et al. 2012). The rationale for that choice is that, in a perfect market, the price of land is the net present value accrued from ownership of the land. Since this value is foregone when the site enters protection, then land value (i.e. acquisition costs) can be considered an appropriate proxy for opportunity costs (Polasky et al. 2001; Adams et al. 2010). Because most land bought for conservation would otherwise be used for agricultural purposes, it follows that agricultural land value should be an appropriate substitute for actual costs of purchasing land for conservation.

This reasoning, however, fails to recognize that there are other, non-market-based dynamics involved in conservation transactions. In Chapter II, we showed that acquisition cost prediction requires more than solely considering alternative land uses and that other socio-economic factors also affect it. In Chapter III, we then defined and characterized one of the dynamics involved that fall outside market economy, taking the form of landowners' propensity to donate part or all of their land to conservation land trusts, instead of selling it at its fair market value. Bargain sales of this type are a common occurrence across the whole contiguous U.S and they have their own spatial pattern that needs to be taken into account when making conservation cost predictions.

C- PERSPECTIVES

Further research is needed to more broadly characterize management costs. We presented a first step in that direction, in Chapter I, but the study of management costs must obviously not stop there. For example, it is important that we investigate such costs accrued over areas protecting different ecosystems. We saw in Chapter II that ecoregions were significant drivers of acquisition costs and in Chapter I that some ecosystem specific activities, such as prescribed burning, were major drivers of management costs in time. In consequence, it would be

interesting to examine to what extent ecoregions also influence management costs. Similar efforts to broaden our understanding of management costs beyond the scope of the study presented in Chapter I are urgently needed. Management costs can sometimes, over the long run, surpass initial acquisition costs and have proven harder for conservation organizations to sustain (Clark 2007). Ultimately, we need to reach a level of understanding of these costs that would allow for large scale, long term prioritization analyses to account for them.

Another aspect of the work presented here that needs further investigation is our understanding of societal support for conservation. Improving the ability of conservation agencies to seize these cost-sharing opportunities is a priority. Resources available for conservation are far too limited and even the best optimization tool cannot stretch conservation dollars to the extent that would be needed (McCarthy et al. 2012). Recognizing the diversity of bottom-up, demand-side driven initiatives and understanding why, how and where they develop would mean being able to capitalize on them when planning for conservation, effectively relaxing budgetary constraints. It could also mean being able to develop strategies aimed at fostering societal participation and involvement, effectively improving the willingness to engage that is necessary for this cost-sharing support to develop.

There is no longer any doubt that accounting for costs in conservation planning is a far superior approach to solely targeting maximal ecological gain (Naidoo et al. 2006; Wilson et al. 2007; Armsworth 2014). The question is now how to do so accurately. When data is not available, how can we correctly estimate costs? When planning conservation at various scales, in both time and space, what are the costs we should consider? When trying to increase available funding, where can we find cost-sharing opportunities and how can we best plan around them? In the current context of all-encompassing ecological crisis, we cannot afford to not find answer to those questions.

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APPENDIX

A- APPENDIX A - CHAPTER I

a. Robustness to buffer size

We fitted our model against a set of landscape-level indicators that were measured within buffer zones of 3 diameters (1, 5 and 10 km) around each site's boundaries. Those values align with recommended distances used to separate unique element occurrences in unsuitable habitat for species survey records (Natureserve 2002).

For each buffer size we ran the same analysis as described in the article. The random component selection identified keeping both management unit and state to be optimal. We then generated all possible models given this model structure, using R-package MuMIn (Barton 2018) and kept all models within $\Delta AICc < 2$ of the best model. Finally, we obtained the averaged model, using AICc weights.

The effect of protected area size, occurrence of prescribed burns and acquisition costs on the costs of managing a protected area are robust to buffer size. They are significantly driving management costs and estimated coefficient for each of those stays mostly unchanged when varying buffer size.

Forest maturity, while still present in at least 50% of the models retained in the set of best models, for each buffer size, are never found to be significant and their 95% confident interval spans 0 in all but one case. Indices for rugosity and visitability are similarly retained but never significant for buffer sizes of 1 km and 10 km.

Across all buffer sizes, site area, acquisition costs and prescribed burn occurrence alone explained on average more than 43% of the variation in management costs. This meant that those 3 parameters (which are not dependent on buffer size) were responsible for 96% of the models' predictive capacity, on average and given the random structure we selected. As a result buffer size was expected to have a relatively small effect on the model selection and averaging process, which is confirmed in Table S.I.1.

b. Endowment calculation

In the result section, we provide an average of the management cost per hectare for our sample. That average was considering all 'sites x year' data points independently, which means that sites that have been protected longer contributed more values to that average. It made sense to think of the average in

that way because our model fit those management costs taken independently and then builds a spatial and temporal dependence structure from there.

When it comes to thinking about endowments, however, it is makes more sense to not let sites that have been protected longer weight more on the average cost per hectare. So for a given site, we calculate management costs per hectare for each year between 2000 and 2014 that the site was protected. We then averaged those yearly values so that we have one single average cost per hectare for each site. When looking at the distribution of those averages across site, we can then calculate the overall average (\$34/ha) and median (\$10/ha).

If building a financial endowment to support the management cost burden associated with protecting a given area, then we want to invest enough so as to be able to pay for management costs every year by withdrawing the earnings only, without decreasing the value of the endowment principal. We assumed an inflation corrected annual rate of return of 4.5% on average (Dahiya & Yermack 2018).

An initial investment of \$233 returning on average 4.5% (compounded annually) will sustainably provide the median management cost value of \$10/ha*year. For the average value of \$34/ha*year, the initial investment will need to be \$790/ha*year. These values are small compared to purchase prices or fair market values of the protected areas.

TABLE S.I. 1: REGRESSION TABLE

Parameters' estimates with 95% confidence intervals for the averaged models obtained from each buffer size (1 km, 5 km and 10 km). *Significance levels: * at 5%, ** at 1% and ***at 0.1% - Greyed out values signal a 95% confidence interval spanning 0.*

Variable	1km	5km	10km
Area (km ²)	0.58±0.21 **	0.69±0.17 ***	0.72±0.20 ***
Rugosity (index)	0.02±0.09	0.46±0.18 **	0.87±0.18
Acquisition Cost (\$)	-0.13±0.06 *	-0.15±0.05 **	-0.16±0.05 **
Visitability (index)	10 E-03 ± 15 E-03	5.2 E-03 ± 1.5 E-03 ***	0.2 E-03±0.5 E-03
Prescribed Burns (yes)	2.80±0.60 ***	2.63±0.59 ***	2.77±0.60 ***
Forest Maturity	<i>mix= 0.35±0.94</i> <i>old= -0.74±0.97</i>	<i>mix= 2.13±1.16</i> <i>old= -0.54±0.92</i>	<i>mix= 0.16±1.09</i> <i>old= -1.01±1.25</i>

c. Relevant survey questions

Site Name: _____

State: _____

Acquisition date: _____

→ Ecological condition and how it has changed since protection:

ecological condition = community composition and how it relates to the desired future forest types at each "block" of a site

(Please refer to the attached aerial photograph of this site)

desired future condition = what you would like that forest to look like in 50 years

- Globally, was the site at ecological condition when acquired? Yes / No
- Globally, how did it change since acquisition? *better / no change / worse*

→ Management costs : definitions

Paid staff-days = total number of work days spent to conduct activities aiming at maintaining or improving the ecological condition of THIS site

e.g. selective forestry, fence maintenance, surveying habitats and species, supervising staff or volunteers at a site, etc.

Other investments: onetime costs and all other costs unambiguously attributable to THIS site

e.g. equipment purchase or replacement

With this in mind, for each management activity, retrieve the following information:

Management activity:

Year it took place:

Number of Staff-days spent on it:

Staff type:

Nb of days spent working on it:

Number of round trips involved:

Did the staff stay overnight? (if yes, enter cost)

Was there a grant for this activity? (if yes, how much)

Was there a contractor involved with this activity? (if yes, enter cost)

Other/Additional costs:

B- APPENDIX B - CHAPTER II

a. Spatial autocorrelation

i- States and Ecoregions as factors

As a first way to account for potential spatial autocorrelation, we decided to test three distinct geographical assemblage patterns as explanatory variables in Model 2. We used states, EPA-1 (8 assemblages across the continental U.S.) and EPA-3 (85 assemblages, see Figure S.I.2). Using EPA-1 ecoregions made the model significantly better than using either states or EPA-3 (the factor "Ecoregions" in Model 2 is EPA-1).

We then built a proximity matrix, calculating pairwise distances between counties and ran a Moran's test on Model 2's residuals weighted by pairwise distances. There was significant (P-value <0.0001) remaining spatial auto-correlation, suggesting the need to adjust the error structure of Model 2 accordingly.

ii- Spatial covariance structure

Generalized least squares methods fit a variance-covariance matrix based on the non-independence of spatial observations (Aitken 1936). We kept the same linear component as in Model 2 and tested five GLS models, each with a different autocorrelation structure (exponential, Gaussian, spherical, linear, and rational quadratic). The rational quadratic correlation structure (Equation S.I.3) was the one that suited our dataset the best (Table S.I.4)

With the rational quadratic structure, and letting d denote the range, the correlation between two observations a distance r apart is:

$$\frac{1}{(1 + (\frac{r}{d})^2)}$$

EQUATION S.I. 2: RATIONAL QUADRATIC STRUCTURE

b. Cross-Validation

i- Spatial cross-validation

We used a repeated k-fold method (with 100 repeats of 10-fold random sets - Kohavi, 1995) to check how well our model, when trained on 90% of the data set only, would predict the remaining 10% data points (out-of-sample cross-

validation). We did the same when comparing predicted values against observed value for the whole dataset at once (in-sample cross-validation)

In both cases we calculated, among other statistics, the root mean squared error (RMSE, Equation S.I.5). RMSE is of the same unit as the response variable and should then be compared the response variable range. In all cases, RMSE was smaller than 12% of the dependent variable range.

$$\sqrt{\frac{\sum_{i=1}^N (\hat{y}_i - y_i)^2}{N}}$$

EQUATION S.I. 3: CROSS-VALIDATION STATISTICS (RMSE)

TABLE S.I. 4: FIVE ERROR STRUCTURES

Model	AICc	ΔAICc
rational quadratic	8916.7	0
original Model 2	9053.4	136.67
exponential	9055.5	138.72
spherical	9055.5	138.72
linear	9055.5	138.72

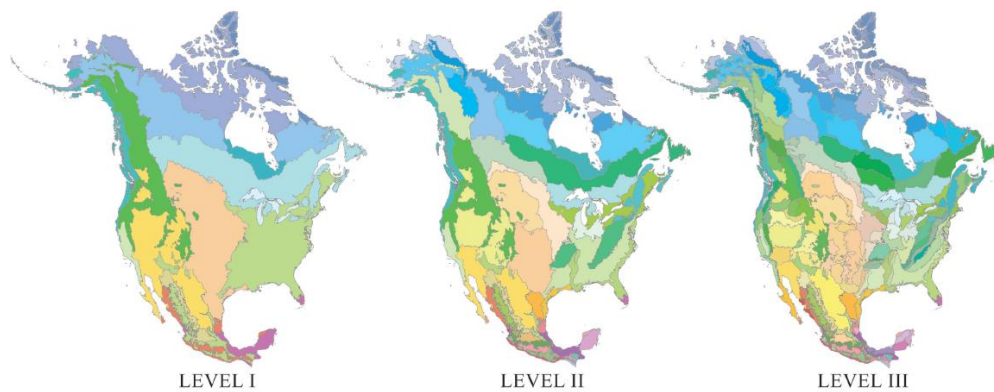


FIGURE S.I. 5: EPA ECOREGIONS, LEVELS 1 TO 3

ii- Temporal cross-validation

Finally, we subdivided the dataset per decade (since it covers 40 years of land purchase, we obtained 4 sub-datasets). We fitted Model 2 on each of these sub-datasets separately. The model was robust and parameter estimates remained consistent over time (Figure S.I.6)

TABLE S.I. 5: MODEL FITTING PER DECADE

*Significance levels: * at 5%, ** at 1% and ***at 0.1%*

	1980's		1990's		2000's		2010's	
	Value	P-val.	Value	P-val.	Value	P-val.	Value	P-val.
(Intercept)	-4.18	*	-4.06	**	-5.61	*	-7.07	**
County.Area	0	.	0	.	0	.	0	.
Average.Parcel.Size	-2.6E-4	***	-9.7E-5	**	-1.8E-4	**	-9.1E-5	***
Ecor. Desert	-	-	-	-	-	-	-	-
Ecor. Mediterranean	0.72	.	0.74	.	0.96	.	0.84	.
Eco. Tropical Forest	1.80	.	1.93	.	1.25	.	1.48	.
Ecor. N. Forests	1.43	**	1.35	.	1.17	**	0.94	***
Ecor. N-W. Forested Mtns	1.02	.	0.54	.	0.85	.	0.78	*
Ecor. Marine W. Coast Forest	0.82	.	1.05	*	1.78	*	1.30	*
Ecor. E. Temperate Forest	0.70	.	0.65	.	1.09	.	0.75	*
Ecor. Great Plains	-7.03	.	-11.72	*	1.83	.	3.50	.
IUCN.Listing	-0.11	.	0.15	*	0.06	*	0.05	*
Elevation	4.7E-4	*	3.4E-4	*	4E-4	*	4.5E-4	*
Urban.Land.Value	0.38	*	0.20	*	0.19	*	0.32	*
Agricultural.Land.Value	0.87	***	0.78	***	0.59	**	0.87	***
Development.Threat	-1.99	.	1.38	.	1.78	.	0.44	.
Education	0.02	**	0.07	***	0.02	*	0.01	***
Population.Density	0.08	*	0.03	*	0.13	**	0.14	*
Median.Income	2.6E-5	.	-1E-6	.	-2.9E-5	.	4E-6	.
Unemployment.Rate	0.23	**	0.38	***	0.11	*	0.19	**
Proportion.Developed	2.17	.	0.29	.	1.84	.	-0.81	.
Proportion.Protected	-6.79	.	1.15	.	1.39	.	-3.39	.

VITA

Diane Le Bouille is a French student who received a master from Paris 6 University in Ecology with a focus on conservation. Her research focuses on conservation planning and optimization. She uses mathematical modeling to better understand and predict costs associated with protected areas, then spatial optimization approaches to inform reserve design and selection. She works directly with conservation organizations and practitioners to help guide conservation practices.