Received: 21 September 2021 | Revised: 18 March 2022 | Accepted: 30 March 2022

DOI: 10.1111/csp2.12697

CONTRIBUTED PAPER

Conservation Science and Practice
A journal of the Society for Conservation Biology
Open Access

WILEY

Spatiotemporal variation in costs of managing protected areas

Diane Le Bouille[1] | Joseph Fargione[2] | Paul R. Armsworth[1]

[1]Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, Tennessee, USA

[2]The Nature Conservancy, North America Region, Minneapolis, Minnesota, USA

Correspondence
Diane Le Bouille, Department of Ecology and Evolutionary Biology, University of Tennessee, Knoxville, 569 Dabney Hall, 1416 Circle Drive, Knoxville, TN 37996, USA.
Email: dlebouille@utk.edu

Funding information
This work was partially supported by the National Science Foundation (NSF, grant number 121142). NSF had no involvement in the study design; in the collection, analysis, and interpretation of data; in the writing of the report; or in the decision to submit the paper for publication.

Abstract

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 management costs or use snapshot estimates, even though they vary through time. We surveyed expenditures on management made over 15 years for 37 protected areas in the Appalachians that were established by a large land trust (TNC). These management costs varied greatly through both time and space. We explored what ecological and socioeconomic characteristics explain this variation. Management costs increased with site area but exhibited 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 expenditures in the years they occur, while acquisition costs were negatively correlated to subsequent expenditures on management. Land managers felt that 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 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.

KEYWORDS

conservation costs, endowment, finance, land trust, management, nature reserve, prescribed fire, protected areas, return-on-investment, stewardship

1 | INTRODUCTION

While buying land to create reserves is a common approach to conservation across the world, it is also costly. Available funding for conservation being 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 (Duke et al., 2013; Laycock et al., 2009; Naidoo & Iwamura, 2007). 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; Murdoch et al., 2007; Polasky et al., 2001). Solving such conservation planning problems accurately can only be done when good quality data is 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 (Bode et al., 2008; Carwardine et al., 2008; Polasky et al., 2001). However, a wide variety of costs are involved 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 land. Management costs can be substantial, sometimes even large enough to outweigh acquisition costs (Armsworth et al., 2011), and adequate resourcing of management costs has been shown to be crucial to achieve conservation goals (Graham et al., 2021; Powlen et al., 2021). Additionally, 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 a 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). As a result, 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. 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 (Carwardine et al., 2012; Chadès et al., 2014; Polasky et al., 2001); 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 (Adams et al., 2019; Kuempel et al., 2018). 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. 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 methodologies. Two contrasting approaches are often taken when seeking to estimate management costs. First, a number of studies seek to estimate ideal budgets that would be needed to achieve a particular ecological goal (Frazee et al., 2003; Lessmann et al., 2019; Wilson et al., 2009). Meanwhile, other studies focus on actual yearly expenditure (Armsworth et al., 2011; Silva et al., 2019), whether or not these are sufficient to achieve protected area management goals. Despite the recurrent nature of management costs, most studies of either type are “snapshot” studies. They look at management costs at one point in time, but across sites that may have been protected for differing amounts of time. For example, Wilson et al. (2009) 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 of the potential temporal variation in costs of managing protected areas through time.

In this article, we describe management costs by focusing on actual expenditures and identify what parameters drive those through both space and time. To do so, we surveyed management costs incurred every year, over 14 years, for 37 protected areas located in the Central and Southern Appalachian Mountains.

2 | METHODS

2.1 | Choice of sites

The Nature Conservancy (hereafter TNC) is the largest conservation nonprofit in the United States, where it owns and manages around 8000 km^2 of land 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, socioeconomics and estimates of the threat of habitat conversion (Conservation Gateway - TNC, 2018). We

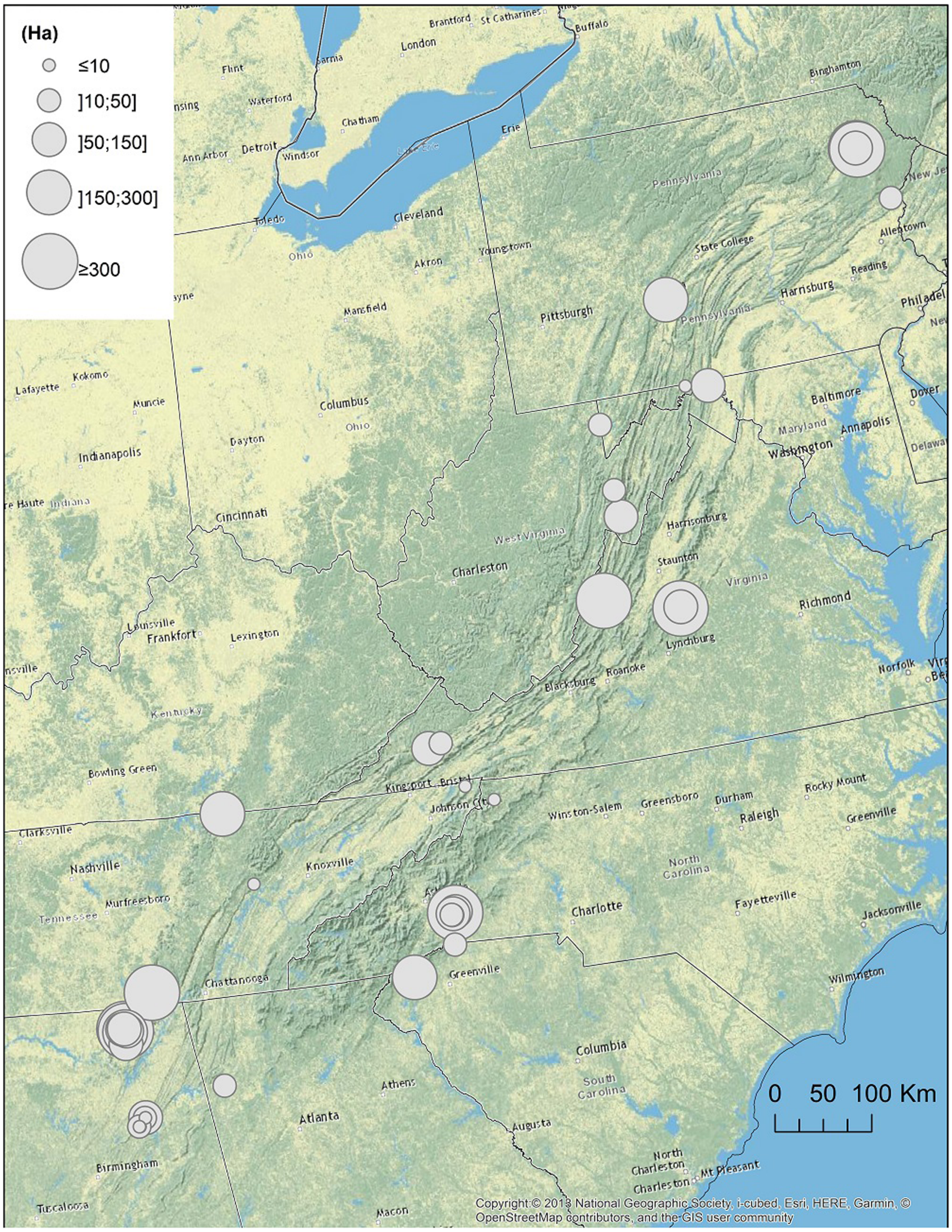

FIGURE 1 Sample of TNC-protected areas: 37 parcels in central and southern Appalachians. Size of circles represents relative size of protected areas (this is for illustration's sake only, all analyses used continuous area)

focused our study on one of those priority regions, which ensured that TNC would own a sufficient number of ecological preserves in the area. The Central and Southern Appalachian region is considered as a hot-spot of biodiversity in the United States (Stein et al., 2000) 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 United States (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 United States.

Our sample of protected lands is comprised of all the areas TNC acquired within the Central and Southern Appalachian Mountains since 2000 that were retained and managed until at least 2014 and for which forest preservation was one of the stated conservation goals. We only included management costs on fully protected areas

Variable	25%	Median	75%
Management cost ($/site/year)	41	242	586
Area (km^2)	0.13	0.69	1.86
Elevation (m)	262	370	577
Rugosity (index)	2.2	2.7	4.2
Acquisition cost ($)	31 k	230 k	630 k
Distance to office (km)	86	157	240
Chapter activity (new land protection per chapter $/year)	0	550 k	1750 k
Agricultural area (%)	4.7	10.1	18.9
Protected area (%)	3.6	13.1	19.2
Visitability	126	254	357

TABLE 1 Model 1's variable distributions: quartiles for continuous variables

and not on conservation easements. This left us with 37 protected areas, which were owned for a minimum of 5 years (9.16 years on average), encompassing nine US 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 three orders of magnitude although many were small (quartiles: 13 – 69 – 186 ha), 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, with individual acquisition prices ranging from 100% donation to more than $6 million.

2.2 | Data acquisition

2.2.1 | 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 were accompanied by a work-sheet form detailing different expenditures; relevant questions can be found in the Appendix S1. We asked, for each site, how much staff-time, whether there were 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. Interestingly, none of the sites received investment specifically targeting invasive removal, although invasive species will have been impacted by some management activities (e.g., herbicide spraying before replanting).

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). Overhead costs are largely organization dependent. Since we worked with TNC only, overhead costs should be broadly comparable across all sites in our sample. In our analyses of what explains variation in these costs, we include a state random factor, which would pick up any remaining variation resulting from differences between various state chapters, (see Section 2.2.3).

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 address the potential bias this could introduce in the Section 3.

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 value. 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 transportations statistics for average fuel efficiency of pick-up trucks in the United States (U.S. Department of Transportation, 2015; U.S. DOE and EPA, 2018) 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 US dollars, using the Consumer Price Index (U.S. Bureau of Labor Statistics, 2019).

Within the analysis, we focused on the management investment made in a given protected parcel of land while including the area of the site as an independent variable in the statistical models. We chose to do this instead of using dollars 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 (Armsworth, 2014; Brett, 2004).

2.2.2 | Explanatory variables

We examined the effect of time on annual management expenditure 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 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 (3 × 3 neighborhood) over the site area with BTM 3.0 ArcGIS Toolbox (Walbridge et al., 2018) because areas located at higher altitudes or whose terrain is more uneven might be more challenging to access and manage, requiring additional equipment and or extra staff time. 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 three diameters (1, 5, and 10 km) around each site's boundaries using Arc GIS Desktop (ESRI, 2015), then we measured the proportion of agricultural land (NatureServe, 2014) and of protected land (USGS Gap Ananlysis 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 Appendix S1 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 like in 50 years' time. We also asked them whether they managed the area with a particular habitat type in mind, and to identify the ecological stage of the forests on site (old growth, in transition or mixed).

Additional details regarding data sources for these covariates and how they were combined are given in Table S1 of the Appendix S1.

2.2.3 | Model fitting, model selection, and model validation

We fitted an initial linear model in R (R Core Team, 2018), complete with all of the explanatory variables listed in Section 2.2.2 (see also Table S1 in Appendix S1), 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 three 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.

There was no significant autocorrelation in our data, neither spatial (Moran test p-value = .5759), nor temporal (Moran test mean p-value = .4774), so we proceeded without an autocorrelation structure for the model. However, 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. We chose a model structure where sites (as management units), states and years

TABLE 2 Regression table best models and average model, with AICc values, R^2_{GLMM} and AICc weights. Parameters' estimates are given with 95% confidence intervals

#	(Intercept)	Area	Rugosity	Acquisition cost	Visitability	Prescribed burns	Forest maturity	Distance office	Agricultural 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*					0.65	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			0.65	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*	0.65	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 = −0401 ± 0.85*			*−0.01 ± 0.02*		0.65	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*				0.65	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*	0.65		

Note: Significance levels:
*at 5%,
**at 1%,
***at 0.1% Values in italic signal a 95% confidence interval spanning 0

were included as random variables. We followed guidelines from Zuur et al. (2009) for model building and selection. Specifically, we first fit the full model as described above, using R-package lme4 (Bates et al., 2014). 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 (ΔAICc >2 with the next best model, using REML estimators). We obtained the following model (Model 1):

$$\begin{aligned}\text{Costs} \sim\ & \text{Area} + \text{Elevation} + \text{Rugosity} + \text{Acquisition.Cost}\\ & + \text{Distance.Office} + \text{Agricultural.Area} + \text{Protected.Area}\\ & + \text{Visitability} + \text{Chapter.Budget} + \text{Prescribed.Burn}\\ & + \text{Forest.Maturity} + \text{Time.Since.Protection}\\ & + \text{Acquisition.Year} + \text{Habitat.Management}\\ & + \text{Ideal.Condition} + (1|\text{Management.Unit}) + (1|\text{State})\end{aligned} \quad (\text{Model1})$$

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 (Bartón, 2018). We compared those using ML estimators and kept all models within ΔAICc < 2 of the best model. This left us with five 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 around normality and homoscedasticity for a model of this type and did not show any sign of autocorrelation, neither spatial nor temporal (see Appendix S1). To examine how much of the predictive capacity of our models was due to particular components, we calculated the R^2_{GLMM} (Nakagawa et al., 2017).

Additional details regarding our statistical analyses can be found in in the Appendix S1 (Section III), including more information about statistical choices we made and the tests we ran to check that model assumptions were met.

2.2.4 | Ecological benefits

We examined covariation of management expenditure with ecological benefit indicators. As a measure of local or onsite ecological benefits, we asked land managers during the survey whether they considered that the site's condition improved, stayed the same or worsened since acquisition. In addition, we calculated two additional indicators that might reflect the ecological importance of protecting different sites to the wider landscape. First, we considered the role protecting the site would have on reducing habitat fragmentation patterns on the wider landscape. As a measure of habitat fragmentation, we used effective mesh size—similar to habitat patch size—within the buffer (Jaeger, 2000) with and without protection of the relevant area. To do so, we used data on protected areas from the PAD-US dataset (USGS Gap Ananlysis Project, 2018). Next, we focused on the site's potential contribution to biodiversity protection. For this, we estimated vertebrate species richness on the sites using modeled species distributions from USGS for 52 species (USGS Gap Analysis Project, 2018). See the Appendix S1 for additional details on those two metrics.

3 | RESULTS

TNC spent $959,000 to manage the sites over a 15-year period. The costs 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 expenditure was in general relatively low; the overall median amount spent when aggregated across both protected areas and years was only around $250 per year (or around $10 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 five sites where that metric was below 50%. Management spendings were highly heterogeneous in both space and time. In terms of spatial variation, average yearly expenditure per protected area ranged from $6/year to $3K/year (or between $0.3 and $435 per hectare per year) across the different protected areas in the study. 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 expenditure (Table 2). Management costs were larger for sites that were bigger, had more rugged terrain, were surrounded by a denser combination of roads and urban areas and cost less to purchase initially. Management costs also peaked whenever a prescribed burn happened. 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.

State, as random component, slightly improved the model compared to accounting only for the management unit (ΔAICc = 2.73), while differences in state chapter budgets did not. This suggests that TNC's internal structure, organized in semi-independent state chapters, might somewhat affect how much is spent on management of different sites (see Appendix S1 for more on the subject of state chapters). Because site managers that we interviewed each worked with a different state chapter, the potential existence of some individual-level bias in managers' recall of past expenditures is also absorbed by this predictor.

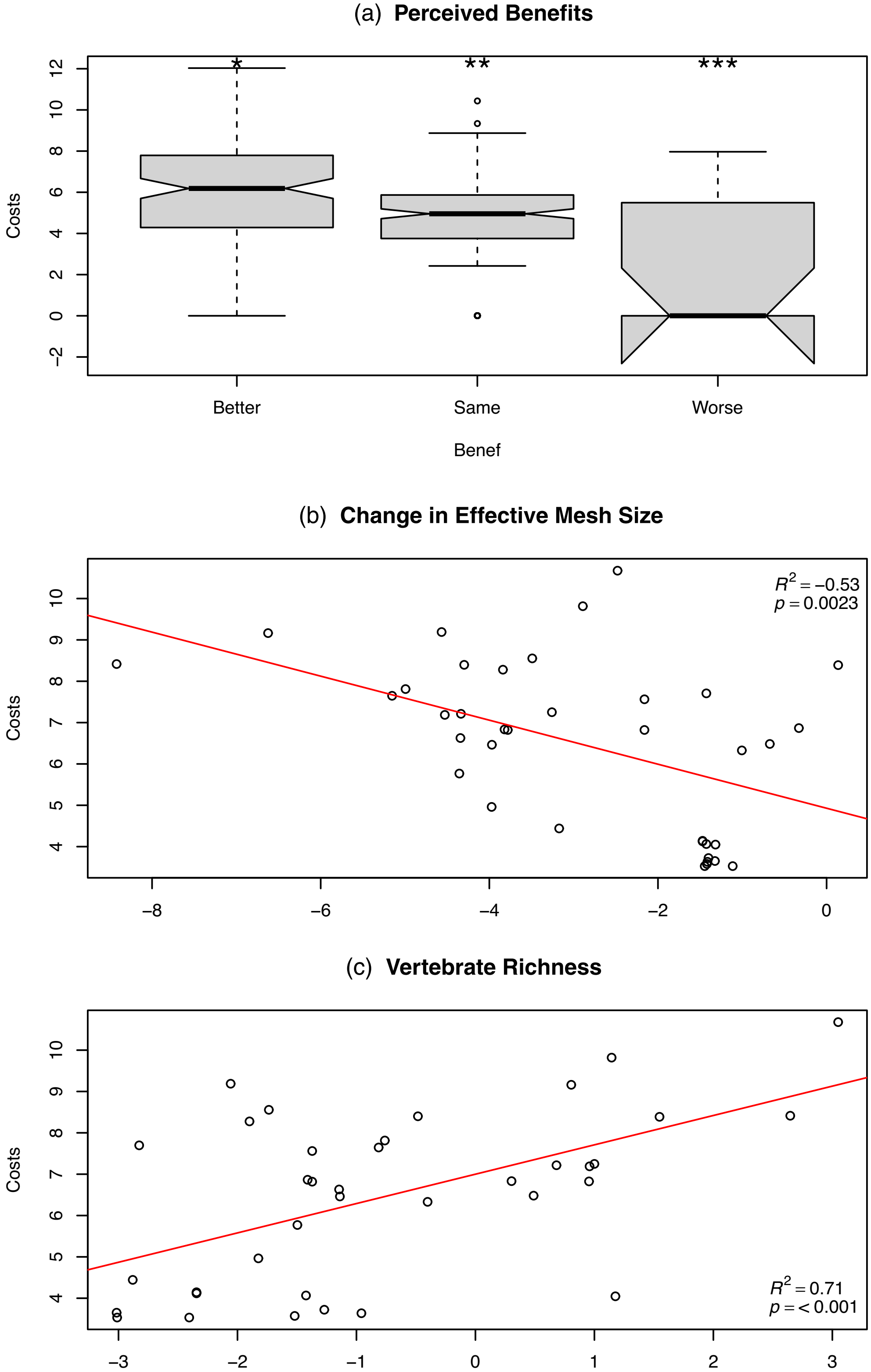

FIGURE 2 Management costs and ecological benefits. Ecological benefits are (a) ecological condition change as perceived by the land manager, (b) change in effective mesh size at the landscape scale due to site acquisition; and (c) vertebrate richness protected by a given site

Local ecological benefit (perceived quality change of the site since time of acquisition, categorized as "better," "same," or "worse") had a significant association with management expenditure: 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 dollars spent (R^2 = -.53, p-value = .0023, 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 expenditure (R^2 = .71, p-value < .001, Figure 2c).

4 | DISCUSSION

Management costs are an important component 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 personal 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.

Expenditure levels on protected areas that we observed (calculated here in terms of direct expenditure and allocations of staff time) were relatively low compared to other studies. For example, \$41 ha^{-1} $year^{-1}$ for a regional public agency in Florida (Dumoulin et al., 2014); \$97 ha^{-1} $year^{-1}$ for protected areas in Brazil (da Silva et al., 2021); \$72 ha^{-1} $year^{-1}$ for invasive control on protected areas in Florida (Iacona et al., 2014). This suggests that TNC may pursue a conservation strategy focused more on protecting greater overall area than on intensive management of areas being protected (Adams et al., 2019; Armsworth et al., 2015). This difference also hints at the fact that there are unmet management needs in the protected areas we examined.

We also observed substantial variation between sites and years in terms of management expenditures. Using a relatively straightforward set of predictor variables, we were able to explain 65% of the variation in management investment, among which 41% is due to the fixed effects alone (using R^2_{GLMM} calculations from Nakagawa et al., 2017). 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 1 ha would increase management costs by less than increasing the size of a small protected area by 1 ha. 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 (Armsworth et al., 2011; Ausden, 2008; Balmford et al., 2003; Kim et al., 2014). Prescribed burns were a strong driver of management investment: 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^2 =.11, p-value = .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.

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 neither simply a function of time elapsed since protection nor are they particularly influenced by the general economic state of the country; indeed, "year" was not retained as a random factor, despite our study encompassing the financial crisis of 2008 and ensuing recession. Others studies have found that the nonprofit sector in general, including TNC and other conservation organizations, was impacted less by the recession than many other sectors (Friesenhahn, 2016; Larson et al., 2014), though TNC reported laying off up to 10% of their staff, organization-wide, in 2009 (Hall, 2009). Using broad measures of economic development to proxy management costs or estimating those as a function of time since protection are common approaches (Armsworth et al., 2011; Bruner et al., 2004; Moore et al., 2004); however our results show that they could be inefficient and generate biased predictions.

Looking at ROIs, 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, while the correlation with species richness on the sites was significantly positive. 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) and those results might point to land managers favoring species-rich sites when allocating management dollars.

Our findings provide a number of insights in regard to land conservation funding. For example, most land trusts fund their management expenses on an endowment basis, and TNC is no exception. All of the protected areas considered in this study were established with an endowment. However, endowment money put aside at time of acquisition was typically pooled for all preserves managed by a given regional office. As such money was freely allocated within a given region. With this financing method, an organization would only require an initial investment of $233 per hectare, assuming an annual rate of return of 4.5% to cover the current expenditure (see Appendix S1 for the methodology and assumption behind that estimate), which means that they could also relatively easily create a larger management budget, with adequate investment at time of purchase. When compared to the fair market value of the protected areas, as detailed in real estate surveyors' estimates provided by TNC, management expenses costed on an endowment basis were on average less than 8% (ranging from 1% to 15%) of the fair market value of buying the sites. That being said, amounts spent varied widely across sites, and in one instance management expenditure even exceeded the site's fair market value. Since variation through time also tends to be large (median CV 108%), conservation organizations tasked with managing these protected areas need a sufficiently flexible budgeting models for any endowments to be able to cope with this variation (Lennox et al., 2017).

Finally, with our study design, we made a number of important choices of which we wish to highlight four. First, we only worked with 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, although it is a common model for conservation land trusts. As noted above, TNC appeared to invest relatively less in site management than other organizations may have done during this time period, likely to allow greater expenditure on additional acquisitions and other activities. Therefore, it will be important to repeat similar designs in other contexts and settings. Second, we focused on management only but conservation organizations face a wider variety of costs, most of which are so far relatively poorly understood. For example, we used a state random variable to pick up broadly ranging variation between state chapters, including potential differences in spending priorities and strategies, as well as other parameters that could influence management spending pattern, such as overhead costs. However, this can only be a temporary solution. More research on the many costs of conservation, their patterns, and what drives them is urgently needed in a context of limited resource and growing conservation needs (Bruner et al., 2004; Naidoo et al., 2006). Third, while we asked managers whether the protected areas were better, the same or worse than when acquired, we did not ask whether there are serious unmet management needs regardless of initial condition. A way to quantify unmet needs would be to ask managers how much more money they would ideally want to be able to spend on a given area. Others have explored those questions before (Frazee et al., 2003; Lessmann et al., 2019; Wilson et al., 2009). We think that those two approaches are complementary and that comparing our results to estimates of ideal budgets would allow better identification of management needs (Lehrer et al., 2019). Fourth, 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 real world, as data on conservation benefits are still scarce (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 fundraising 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—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.

ACKNOWLEDGMENT

We thank TNC staff for participating in the survey and advising on the overall design, and G. Iacona for helpful comments.

CONFLICT OF INTEREST

There are no conflicts of interest reported for any of the authors.

AUTHORS' CONTRIBUTIONS

Diane Le Bouille led the data collection and analysis and wrote the paper; Joseph Fargione facilitated contact with land managers, provided feedback on the design and edited the paper; Paul R. Armsworth provided feedback on the design and analysis, and significantly edited each draft of the manuscript.

DATA AVAILABILITY STATEMENT

We can only release aggregated data due to the potential for reidentification of survey subjects. This means we are unable to make the raw data from surveys publicly available. We have created a repository at https://github.com/dlebouille/Le-Bouille-et-al.CSP-2022 containing supplemental material, including the code and email templates that were used in the surveys, as well as the full survey questionnaire. We also included a means to estimate predicted costs of individual protected areas using the results of our model fitting in this repository.

ORCID

Diane Le Bouille https://orcid.org/0000-0002-3373-7282
Joseph Fargione https://orcid.org/0000-0002-0636-5380
Paul R. Armsworth https://orcid.org/0000-0003-0918-0573

REFERENCES

Adams, V. M., Iacona, G. D., & Possingham, H. P. (2019). Weighing the benefits of expanding protected areas versus managing existing ones. *Nature Sustainability*, *2*, 404–411. https://doi.org/10.1038/s41893-019-0275-5

Ando, A., Camm, J., Polasky, S., & Solow, A. (1998). Species distributions, land values, and efficient conservation. *Science*, *279* (5359), 2126–2128. https://doi.org/10.1126/science.279.5359.2126

Armsworth, P. R. (2014). Inclusion of costs in conservation planning depends on limited datasets and hopeful assumptions. *Annals of the New York Academy of Sciences*, *1322*(1), 61–76. https://doi.org/10.1111/nyas.12455

Armsworth, P. R., Cantú-Salazar, L., Parnell, M., Davies, Z. G., & Stoneman, R. (2011). Management costs for small protected areas and economies of scale in habitat conservation. *Biological Conservation*, *144*(1), 423–429. https://doi.org/10.1016/j.biocon.2010.09.026

Armsworth, P. R., Jackson, H. B., Cho, S. H., Clark, M., Fargione, J. E., Iacona, G. D., Kim, T., Larson, E. R., Minney, T., & Sutton, N. A. (2018). Is conservation right to go big? Protected area size and conservation return-on-investment. *Biological Conservation*, *225*, 229–236. https://doi.org/10.1016/j.biocon.2018.07.005

Armsworth, P. R., Larson, E. R., Jackson, S. T., Sax, D. F., Simonin, P., Blossey, B., Green, N., Klein, M. L., Lester, L., Ricketts, T. H., Runge, M. C., & Shaw, M. R. (2015). Are conservation organizations configured for effective adaptation to global change? *Frontiers in Ecology and the Environment*, *13*(3), 163–169. https://doi.org/10.1890/130352

Ausden, M. (2008). *Habitat management for Conservation*. Oxford University Press. https://doi.org/10.1093/acprof:oso/9780198568728.001.0001

Balmford, A., Gaston, K. J., Blyth, S., James, A., & Kapos, V. (2003). Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs. *Proceedings of the National Academy of Sciences of the United States of America*, *100*(3), 1046–1050. https://doi.org/10.1073/pnas.0236945100

Bartón, K. (2018). *MuMIn: multi-model inference*.

Bates, D., Maechler, M., Bolker, B. & Walker, S. (2015). Fitting linear mixed-effects models using lme4. *Journal of Statistical Software 67*(1), 1–48. https://doi.org/10.18637/jss.v067.i01

Bode, M., Wilson, K. A., Brooks, T. M., Turner, W. R., Mittermeier, R. A., McBride, M. F., Underwood, E. C., & Possingham, H. P. (2008). Cost-effective global conservation spending is robust to taxonomic group. *Proceedings of the National Academy of Sciences of the United States of America*, *105*(17), 6498–6501. https://doi.org/10.1073/pnas.0710705105

Brett, M. T. (2004). When is a correlation between non-independent variables "spurious"? *Oikos*, *105*(3), 647–656. https://doi.org/10.1111/j.0030-1299.2004.12777.x

Bruner, A. G., Gullison, R. E., & Balmford, A. (2004). Financial costs and shortfalls of managing and expanding protected-area systems in developing countries. *Bioscience*, *54*(12), 1119–1126. https://doi.org/10.1641/0006-3568(2004)054[1119:FCASOM]2.0.CO;2

Carwardine, J., O'Connor, T., Legge, S., Mackey, B., Possingham, H. P., & Martin, T. G. (2012). Prioritizing threat management for biodiversity conservation. *Conservation Letters*, *5*(3), 196–204. https://doi.org/10.1111/j.1755-263X.2012.00228.x

Carwardine, J., Wilson, K. A., Ceballos, G., Ehrlich, P. R., Naidoo, R., Iwamura, T., Hajkowicz, S. A., & Possingham, H. P. (2008). Cost-effective priorities for global mammal conservation. *Proceedings of the National Academy of Sciences of the United States of America*, *105*(32), 11446–11450. https://doi.org/10.1073/pnas.0707157105

Chadès, I., Chapron, G., Cros, M. J., Garcia, F., & Sabbadin, R. (2014). MDPtoolbox: A multi-platform toolbox to solve stochastic dynamic programming problems. *Ecography (Cop.)*, *37*(9), 916–920. https://doi.org/10.1111/ecog.00888

Clark, S. (2007). *A Field Guide to Conservation Finance*. Island Press.

Conservation Gateway - The Nature Conservancy. (2018). *Ecoregional assessment [WWW Document]*. https://www.conservationgateway.org/ConservationPlanning/SettingPriorities/EcoregionalAssessment/Pages/ecoregional-assessment.aspx

Cook, C. N., Pullin, A. S., Sutherland, W. J., Stewart, G. B., & Carrasco, L. R. (2017). Considering cost alongside the effectiveness of management in evidence-based conservation: A systematic reporting protocol. *Biological Conservation*, *209*, 508–516. https://doi.org/10.1016/j.biocon.2017.03.022

da Silva, J. M. C., Dias, T. C. A. d. C., da Cunha, A. C., & Cunha, H. F. A. (2021). Funding deficits of protected areas in Brazil. *Land Use Policy*, *100*, 104926. https://doi.org/10.1016/J.LANDUSEPOL.2020.104926

Duke, J. M., Dundas, S. J., & Messer, K. D. (2013). Cost-effective conservation planning: Lessons from economics. *Journal of Environmental Management*, *125*, 126–133. https://doi.org/10.1016/j.jenvman.2013.03.048

Dumoulin, C. E., Macmillan, T., Stoneman, R., & Armsworth, P. R. (2014). Locating human resources to reduce the cost of managing networks of protected areas. *Conservation Letters*, *7*(6), 553–560. https://doi.org/10.1111/conl.12115

Environmental Systems Research Institute (ESRI). (2015). *ArcGIS desktop*.

Frazee, S. R., Cowling, R. M., Pressey, R. L., Turpie, J. K., & Lindenberg, N. (2003). Estimating the costs of conserving a biodiversity hotspot: A case-study of the cape floristic region, South Africa. *Biological Conservation*, *112*(1-2), 275–290. https://doi.org/10.1016/S0006-3207(02)00400-7

Friesenhahn, E. (2016). Nonprofits in America: New research data on employment, wages, and establishments. *Monthly Labor Review*. https://doi.org/10.21916/mlr.2016.9

Graham, V., Geldmann, J., Adams, V. M., Grech, A., Deinet, S., & Chang, H. C. (2021). Management resourcing and government transparency are key drivers of biodiversity outcomes in Southeast Asian protected areas. *Biological Conservation*, *253*, 108875. https://doi.org/10.1016/j.biocon.2020.108875

Hall, H. (2009). Nature conservancy lays off 10% of its staff. *Chronicle of Philanthropy*.

Iacona, G. D., Price, F. D., & Armsworth, P. R. (2014). Predicting the invadedness of protected areas. *Diversity and Distributions*, *20*(4), 430–439. https://doi.org/10.1111/DDI.12171

Iacona, G. D., Sutherland, W. J., Mappin, B., Adams, V. M., Armsworth, P. R., Coleshaw, T., Cook, C., Craigie, I., Dicks, L. V., Fitzsimons, J. A., McGowan, J., Plumptre, A. J., Polak, T., Pullin, A. S., Ringma, J., Rushworth, I., Santangeli, A., Stewart, A., Tulloch, A., … Possingham, H. P. (2018). Standardized reporting of the costs of management interventions for biodiversity conservation. *Conservation Biology*, *32*(5), 979–988. https://doi.org/10.1111/cobi.13195

Jaeger, J. A. G. (2000). Landscape division, splitting index, and effective mesh size: New measures of landscape fragmentation. *Landscape Ecology*, *15*, 115–130.

Jenkins, C. N., Van Houtan, K. S., Pimm, S. L., & Sexton, J. O. (2015). US protected lands mismatch biodiversity priorities. *Proceedings of the National Academy of Sciences of the United States of America*, *112*(16), 5081–5086. https://doi.org/10.1073/pnas.1418034112

Kim, T., Cho, S. H., Larson, E. R., & Armsworth, P. R. (2014). Protected area acquisition costs show economies of scale with area. *Ecological Economics*, *107*, 122–132. https://doi.org/10.1016/j.ecolecon.2014.07.029

Kuempel, C. D., Adams, V. M., Possingham, H. P., & Bode, M. (2018). Bigger or better: The relative benefits of protected area network expansion and enforcement for the conservation of an exploited species. *Conservation Letters*, *11*, e12433(3), https://doi.org/10.1111/conl.12433

Kujala, H., Lahoz-Monfort, J. J., Elith, J., & Moilanen, A. (2018). Not all data are equal: Influence of data type and amount in spatial conservation prioritisation. *Methods in Ecology and Evolution*, *9*(11), 2249–2261. https://doi.org/10.1111/2041-210X.13084

Larson, E. R., Boyer, A. G., & Armsworth, P. R. (2014). A lack of response of the financial behaviors of biodiversity conservation nonprofits to changing economic conditions. *Ecology and Evolution*, *4*(23), 4429–4443. https://doi.org/10.1002/ece3.1281

Laycock, H., Moran, D., Smart, J., Raffaelli, D., & White, P. (2009). Evaluating the cost-effectiveness of conservation: The UK biodiversity action plan. *Biological Conservation*, *142*(12), 3120–3127. https://doi.org/10.1016/j.biocon.2009.08.010

Lehrer, D., Becker, N., & Bar, P. (2019). The drivers behind nature conservation cost. *Land Use Policy*, *89*, 104222. https://doi.org/10.1016/J.LANDUSEPOL.2019.104222

Lennox, G. D., Fargione, J., Spector, S., Williams, G., & Armsworth, P. R. (2017). The value of flexibility in conservation financing. *Conservation Biology*, *31*(3), 666–674. https://doi.org/10.1111/cobi.12771

Lessmann, J., Fajardo, J., Bonaccorso, E., & Bruner, A. (2019). Cost-effective protection of biodiversity in the western Amazon. *Biological Conservation*, *235*, 250–259. https://doi.org/10.1016/j.biocon.2019.04.022

Mockrin, M.H., Lilja, R.L., Weidner, E., Stein, S.M. & Carr, M.A. (2014). Private forests, housing growth and America' s water supply: A report from the forests on the edge and forests to faucets projects, General Technical Report. https://doi.org/10.2737/RMRS-GTR-327

Moore, J., Balmford, A., Allnutt, T., & Burgess, N. (2004). Integrating costs into conservation planning across Africa. *Biological Conservation*, *117*(3), 343–350. https://doi.org/10.1016/j.biocon.2003.12.013

Murdoch, W., Polasky, S., Wilson, K. A., Possingham, H. P., Kareiva, P., & Shaw, R. (2007). Maximizing return on investment in conservation. *Biological Conservation*, *139*(3-4), 375–388. https://doi.org/10.1016/j.biocon.2007.07.011

Naidoo, R., Balmford, A., Ferraro, P. J., Polasky, S., Ricketts, T. H., & Rouget, M. (2006). Integrating economic costs into conservation planning. *Trends in Ecology & Evolution*, *21*(12), 681–687. https://doi.org/10.1016/j.tree.2006.10.003

Naidoo, R., & Iwamura, T. (2007). Global-scale mapping of economic benefits from agricultural lands: Implications for conservation priorities. *Biological Conservation*, *140*(1-2), 40–49. https://doi.org/10.1016/j.biocon.2007.07.025

Nakagawa, S., Johnson, P. C. D., & Schielzeth, H. (2017). The coefficient of determination R 2 and intra-class correlation coefficient from generalized linear mixed-effects models revisited and expanded. *Journal of Royal Society Interface*, *14*(134). https://doi.org/10.1098/rsif.2017.0213

NASA-JPL. (2013). *NASA shuttle radar topography Mission global 1 arc second number*. Nasa Lp Daac. https://doi.org/10.5067/MEaSUREs/SRTM/SRTMGL1.003

NatureServe (2014). Terrestrial ecological systems of the conterminous United States. Version 3.0. https://www.natureserve.org/products/terrestrial-ecological-systems-united-states

Polasky, S., Camm, J. D., & Garber-Yonts, B. (2001). Selecting biological reserves cost-effectively: An application to terrestrial vertebrate conservation in Oregon. *Land Economics*, *77*(1), 68–78. https://doi.org/10.2307/3146981

Powlen, K. A., Gavin, M. C., & Jones, K. W. (2021). Management effectiveness positively influences forest conservation outcomes in protected areas in Mexico. *Biological Conservation*, *260*, 109192. https://doi.org/10.1016/J.BIOCON.2021.109192

Silva, J. M. C. d., Castro Dias, T. C. A. d., Cunha, A. C. d., & Cunha, H. F. A. (2019). Public spending in federal protected areas in Brazil. *Land Use Policy*, *86*, 158–164. https://doi.org/10.1016/j.landusepol.2019.04.035

Stein, B. A., Kutner, L. S., & Adams, J. S. (Eds.). (2000). *Precious heritage: The status of biodiversity in the United States*. Oxford University Press. https://doi.org/10.1046/j.1526-100x.2001.94017.x

R Core Team. (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

U.S. Bureau of Labor Statistics (2019). CPI (consumer price index) - Inflation calculator. https://www.bls.gov/data/inflation_calculator.htm

U.S. Census Bureau (2015). 2015 TIGER/Line Shapefiles (machine-readable data files). www.census.gov/geo/maps-data/data/tiger-line.html

U.S. Department of Transportation (2015). Average fuel efficiency of U.S. light duty vehicles, national transportation statistics. https://www.bts.gov/content/average-fuel-efficiency-us-light-duty-vehicles

U.S. DOE, EPA (2018). Fuel economy - pickup trucks. https://www.fueleconomy.gov/feg/PowerSearch.do?action=noform&year=2000&mclass=Pickup%20Trucks

U.S. Energy Information Administration (2018). Gasoline and diesel fuel report. https://www.eia.gov/petroleum/gasdiesel/

USGS Gap Analysis Project. (2018). *Gap analysis project species range maps*. https://doi.org/10.5066/F7Q81B3R

USGS Gap Ananlysis Project (2018). Protected areas database of the United States (PADUS) version 2.0. https://doi.org/10.5066/P955KPLE

van Wilgen, B. W., Fill, J. M., Govender, N., & Foxcroft, L. C. (2017). An assessment of the evolution, costs and effectiveness of alien plant control operations in Kruger National Park, South Africa. *NeoBiota*, *35*, 35–59. https://doi.org/10.3897/neobiota.35.12391

Walbridge, S., Slocum, N., Pobuda, M., & Wright, D. J. (2018). Unified geomorphological analysis workflows with benthic terrain modeler. *Geoscience*, *8*(3), 94. https://doi.org/10.3390/geosciences8030094

Wilson, K. A., Cabeza, M., & Klein, C. J. (2009). Fundamental concepts of spatial conservation prioritization. In A. Moilanen, K. A. Wilson & H. P. Possingham, *Spatial conservation prioritization*. (16–27). Oxford University Press.

Zuur, A. F., Ieno, E. N., Elphick, C. S., Walker, N. J., Saveliev, A. A., & Smith, G. M. (2009). *Mixed Effect Models and Extensions in Ecology with R*. Springer. https://doi.org/10.1007/978-0-387-87458-6

SUPPORTING INFORMATION

Additional supporting information may be found in the online version of the article at the publisher's website.

How to cite this article: Le Bouille, D., Fargione, J., & Armsworth, P. R. (2022). Spatiotemporal variation in costs of managing protected areas. *Conservation Science and Practice*, *4*(6), e12697. https://doi.org/10.1111/csp2.12697