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Conservation easements target high quality lands but do not increase their quality

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ABSTRACT

Conservation programs targeted at private lands are essential for conserving biodiversity and mitigating and adapting to the effects of climate change. Private land conservation programs typically focus on maximizing acres enrolled, but their outcomes are less studied. We used a counterfactual approach to measure the efficacy of private land protection investments in a high-value conservation region of the western United States, where private agricultural lands provide critical habitats that are not well-protected by public protected areas, but are highly vulnerable to development. We used difference-in-differences panel regressions and annual time series maps of land cover and mesic habitat quality derived from satellite imagery to measure whether conservation easements a) were placed on private lands of higher conservation quality compared to non-easements, and b) improved ecosystem condition after implementation. We found that conservation easements targeted private lands that are less developed and have more healthy ecosystems compared to non-easements. However, we found no evidence that after implementation easements were consistently less likely to be developed, or led to improved mesic ecosystem conditions. Our findings suggest that easements are being placed on high quality lands for conservation, but that they may be a missed opportunity for conservation because conservation and restoration are not always explicit goals of conservation easements, and thus they are not leading to ecosystem improvement after implementation. Through this analysis, we demonstrate the value of low-cost satellite monitoring protocols and statistical impact evaluation to assess conservation actions implemented on private lands.

1. Introduction

For decades, governments and conservation organizations globally have made concerted efforts to increase protected area networks in an effort to conserve global biodiversity (Joppa and Pfaff, 2009). Initiatives such as the Aichi targets set by the Convention for Biological Diversity to protect 17 % of the terrestrial surface (Obura et al., 2021) and the '30 by 30' initiative meant to conserve 30 % of ecosystems by 2030 aim to protect enough of the earth to maintain the biodiversity that sustains it (Belote et al., 2021). However, biodiversity and ecosystem functions that protected areas intend to protect continue to decline (Butchart et al., 2015; Drescher and Brenner, 2018). Limited success can be attributed to biases in the way protected areas are designated relative to economic incentives, as most often they occur in undesirable locations where agricultural suitability is low (Venter et al., 2018). Further, protected areas are rarely designated to maximize habitats for species at risk of extinction (Pimm et al., 2014). Common targets aim to increase the area of terrestrial protected areas, but increases in protected areas do not effectively result in more habitats conserved for threatened species due to the inherent biases in siting process (Venter et al., 2014). Furthermore, growing human populations are appropriating more land and resources, which exacerbates conservation challenges (Ehrlich, 1995).The massive investment in the public protected area network has not been sufficient to protect biodiversity and strategies beyond public protected areas are integral in maintaining biodiversity and ecosystem functions (Butchart et al., 2015; Woodley et al., 2012).

Conservation on private lands has great potential to fill important gaps in the protected area network. One important way that private lands could provide important contributions to the protected area network is that private lands, unlike public lands, tend to be on the most

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productive sites. Public lands have typically been designated "high and far", that is, in areas that are relatively undesirable for humans and wildlife (Butchart et al., 2015; Joppa and Pfaff, 2009). Conserving only public lands excludes some species of concern and threatened ecosystems entirely (Graves et al., 2019; Pimm et al., 2014). Strategies that focus on private lands have shown promise for biodiversity through targeted habitat conservation in key biomes with disproportionate importance for ecological functions, but relatively low representation on public lands (Palfrey et al., 2022). Protection of these relatively intact rare, but disproportionally important systems, are key to maintaining already limited landscape level ecological functions. However, whether private land protection strategies are systematically targeting the highest quality ecosystems has not been determined.

A second important way in which private land protection could contribute to the protected area network is if these protections lead to ecosystem improvement following their implementation. However, there are major challenges that come with incorporating conservation goals into private land protection strategies. First, protection of private lands tends to vary in the ways it is implemented and its specific goals, because agreements are negotiated with individual landowners for specific parcels of land. Therefore, protected parcels tend to be smaller and more spread out than public protected areas, implemented at different times, and managed by different people. Another challenge is that there are typically two desired outcomes of private land protection that are not always in balance. The most explicit outcome is to protect private land from more intensive land uses, such as commercial, industrial, or residential development (Braza, 2017; Rissman, 2013). The second desired outcome is typically less explicit – that of ecosystem conservation and improvement (Rissman et al., 2007). This not only creates inconsistency among private land protection in terms of the desired conservation outcomes, but it also makes monitoring outcomes and enforcing rules difficult. Though the outcomes of restricting development are meant to maintain intact ecosystems and landscapes, it has not been determined whether private land protection efforts effectively conserve ecologically sensitive habitats.

Conservation easements (hereafter easements) represent an excellent opportunity to quantify the extent to which private land protection is fulfilling its potential to complement public protected area networks. Easements are a popular and theoretically well-designed strategy to conserve sensitive ecosystems on private land by restricting development (Braza, 2017; Fishburn et al., 2009). In particular, legal agreements between non-governmental organizations (NGOs; e.g. land trusts, conservation organizations) and landowners have become increasingly popular relative to government interventions due to lack of trust and patience with centralized regulatory authorities and the rising cost of land and its management (Merenlender et al., 2004). Easement holders acquire easements at less than market value and an agreement is drawn based on agreed upon land uses, typically limiting residential, commercial, and industrial development. The landowner remains the steward of the land, while operating within the confines of the agreement. Often, these areas are designated in the interest of providing refugia for wildlife and maintaining ecosystem functions, but are also commonly used to protect agricultural lands. In both instances, easements are a tool to stave off development, although many protection arrangements allow for development to some degree (Rissman et al., 2007; Rissman and Sayre, 2012). Despite the growing prominence of easements, rigorous analyses of their efficacy are limited. Nolte et al. (2019) demonstrate conservation easements on private lands to be slightly effective for reducing the likelihood of conversion of forested systems to residential areas in New England, and Byrd et al. (2009) show that conservation easements may change development patterns, but not overall habitat loss in blue oak woodlands of California. Easements have been found to occur near streams, in riparian areas and floodplains, theoretically increasing ecological resilience and overall landscape connectivity (Fremier et al., 2015). However, easements do not always connect well with other protected areas (Stoms et al., 2009), nor are necessarily placed optimally for landscape connectivity and conservation of biodiversity (Graves et al., 2019). These and other initial explorations elicit the need for more analyses of the efficacy of easements as an institution for maintaining ecosystem functions and integrity, particularly given the monetary investments associated with them (Kiesecker et al., 2007).

The American West is a region where private land conservation has great potential to contribute to societal goals of biodiversity conservation and ecosystem functions. Though public lands dominate in the Western United States, they are unable to support conservation goals on their own due to the disproportionate amount of biodiversity and ecologically sensitive habitats found on private lands (Maestas et al., 2001; Woodley et al., 2012). The West has been experiencing population increases in recent years which lead to concomitant increases in demand for residential and commercial development (Jones et al., 2019). Exurban development occurring in rural areas is transforming the West, often leading to adverse ecological conditions (Goldstein et al., 2011; Hansen et al., 2002; Maestas et al., 2001). Development typically occurs in easy to access, relatively flat valley bottom areas of riverscapes where water is readily available (Ahmed and Jackson-Smith, 2019). Protection via preservation and conservation of privately-owned areas in drylands that encompass areas of high biodiversity, productivity, and water is necessary for maintaining landscape level functions and mitigating the effects of climate change (Donnelly et al., 2016; Robinson et al., 2019).

Our goal in this study was to quantify the effectiveness of private land protection as a conservation tool. We conducted our study in the High Divide region of Idaho and Montana, USA, which has made major investments in easements as the dominant mode of private land protection. The region is representative of the problems facing the Western United States more generally: increasing population, increasing aridity, and a shift from agricultural to amenity driven economies (Jones et al., 2019). These changes all exacerbate the effects of climate change and the increasing uncertainty around water availability, particularly in the driest portions of the water year. We used remote sensing derived time series datasets of valley bottom development and mesic vegetation, an indicator of ecologically available water and thus a proxy for biodiversity, habitat availability, and riparian health, along with counterfactual analyses to ask: Do easements target private lands with limited development and high quality mesic ecosystems? Do easements reduce development and lead to better mesic ecosystem outcomes compared to non-easement private lands? We also demonstrate the necessity of using counterfactual approaches vs simple comparisons when evaluating private land protection strategies, and we show the utility of satellite-derived indicators for overcoming the substantial challenges of measuring outcomes on private lands.

2. Methods

2.1. Study area

The High Divide is a mountainous region of eastern Idaho and western Montana that stretches between the Greater Yellowstone Ecosystem, the Salmon-Selway wilderness area, and the Crown of the Continent; three relatively intact ecosystems in Western North America (Belote et al., 2016) (Fig. 1). Precipitation in the region is dominated by winter snowfall, with high elevation snowmelt being the major source of water inputs, particularly during summer months. Typical of the Western United States, ~60 % of the land in this region is public, managed by federal and state agencies (Graves et al., 2019; Jones et al., 2019). The landscape is transitioning from more of an agricultural to amenity driven economy (Winkler et al., 2007), with many outdoor recreation opportunities in all seasons. Private lands, typically found in low-lying valley bottoms, were historically used for agricultural purposes, but increasing exurban development further threatens the functioning of this landscape as a conduit between protected areas (Brown et al., 2005; Carroll et al., 2012). The High Divide embodies threats to mesic ecosystems found largely on private lands throughout the West, where their

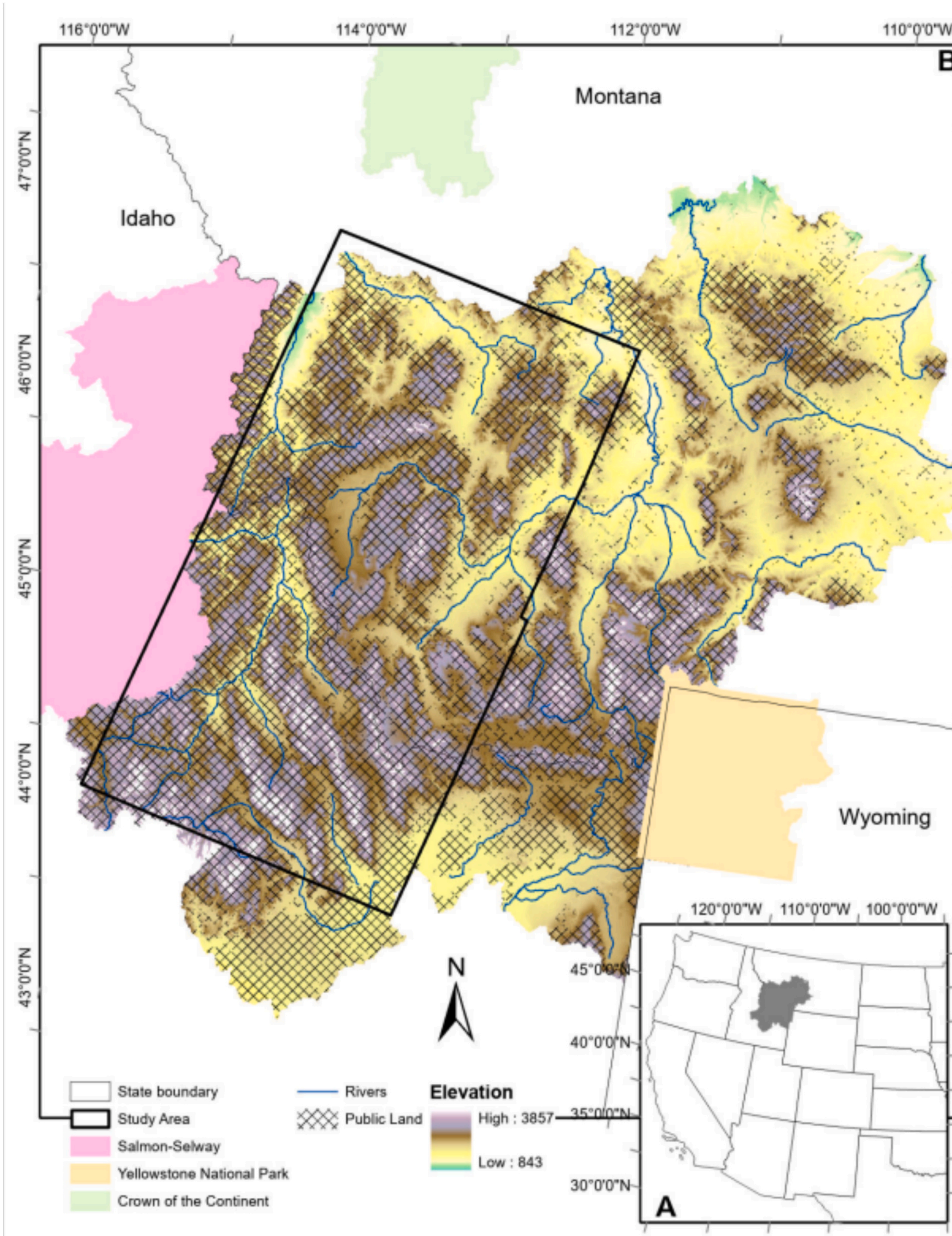

Fig. 1. A) Location of the High Divide region. B) Elevation (m), public land, and rivers within the High Divide boundary; surrounding intact ecosystems; relative location of the study area.

conservation has implications not only for iconic species such as salmonids (McClure et al., 2008) and sage grouse (Donnelly et al., 2016), but for landscape level functions as well.

Riparian ecosystems, though only occupying ~2 % of the landscape, provide habitat and refugia for ~70 % of wildlife species, and are disproportionately important for the success of landscape and biodiversity conservation efforts (Poff et al., 2012). Vegetation communities specific to wet meadows, riparian corridors, and wetlands (hereafter mesic ecosystems) are indicative of available water (Kolarik et al., 2023) and known to be of critical importance for threatened and iconic species in the West such as sage grouse (Donnelly et al., 2016) and elk (Barker et al., 2019). Warming temperatures, variable precipitation inputs, and riverscapes that lack complexity due to land use and land cover changes lead to unsuitable riparian and instream habitats, and lower water availability in the hottest, driest part of the water year (Bouwes et al., 2016; Cluer and Thorne, 2014; Dauwalter and Walrath, 2018; Poole and Berman, 2001). Donnelly et al. (2016) demonstrate that mesic ecosystems in the semi-arid American West largely occur on private lands. It is clear that private land protection has potential to play a significant role in effective mesic ecosystem conservation and climate change mitigation, but it is unclear whether high quality mesic areas are being effectively targeted, restored, and conserved.

Development in the Intermountain West often comes at the expense of healthy riverscapes due to the topographic complexity. Riverscapes found in the relatively low lying valley bottom areas are the conduits through which water exits the system. Agricultural and infrastructural development has made these conduits more efficient and deeply incised channels move water swiftly from the system, leaving historical floodplains parched and increasingly disconnected from the main channel, particularly during the late summer months. This disconnection leads to degraded riverscapes, decreased opportunities for infiltration, unsuitable habitats, lower water tables, and reduced groundwater retention (Pollock et al., 2014). With the bulk of private lands and intensive anthropogenic activities restricted to valley bottoms, conservation initiatives that seek to maintain the ecological integrity and connectivity within these landscapes must focus on limiting the development of these sustaining mesic ecosystems.

We use these context specific details of the Intermountain West to frame our hypothesized causal pathways and the associated theory of change (Fig. S1). First, when prescribing restoration treatments, environmental assessments are considered along with the livelihoods of the people living on the landscape, for instance farmers and ranchers. Much of the conservation discourse in this region and arena is centered around working lands capable of sustaining wildlife, with lofty goals of supporting both endeavors equally (Bestelmeyer and Briske, 2012; Charnley et al., 2020; Runge et al., 2019). The strategic habitat restoration targets developed by conservation organizations and collaborative groups typically involve purchasing the development rights on land from private owners to restrict development (i.e. easement implementation) and altering land use practices (i.e. strategic irrigation and grazing plans). Ideally, all activities allow for flexible management to address any potential problems (Charnley et al., 2018). The conservation actions taken by these groups are meant to lead to restored habitats that increase connectivity and protect land from development that may further degrade landscape level functions, leaving behind a flexible social-ecological system well positioned to support livelihoods on working lands and the ecosystem functions that sustain the system.

2.2. Conservation status

We assigned three categories of land tenure: a) conservation easements on private land, b) non-easements, or non-protected private land, and c) public lands. We used the conservation easements dataset curated by Graves et al. (2019) specifically for the High Divide. Since the National Conservation Easements Database contains only voluntarily contributed easement locations, the authors supplemented this dataset with information about unreported easements from personal communication with land trusts located within the High Divide. We subset these data to represent only easements implemented before 2016, allowing for ample time following implementation to measure any impacts. To characterize whether conservation was a stated goal of easements, we used the updated USGS PAD-US 2.0 to compile the easement holder and reported GAP status for each easement. Graves et al. (2019) also determined public lands boundaries and their conservation status (GAP status) using the US Geological Survey GAP Protected Area Database of the US (USGS PAD-US 1.4). GAP status is defined by four categories: GAP 1 - Areas with permanent protection from conversion of natural land cover and managed for biodiversity where natural disturbances are allowed to proceed; GAP 2 - Areas with permanent protection from conversion of natural land cover and managed for biodiversity where natural disturbance is suppressed; GAP 3 - Areas protected from land cover conversion to protect federally listed endangered and threatened species but subject to extractive and recreational uses; GAP 4 - Areas with no known mandate for protection. We also compiled information about the easement holder for each easement. We visited the web pages for each easement holder and combed through their mission statements for evidence that biodiversity, wildlife, habitat, wetlands, rivers, and riparian ecosystem improvements are part of the mission of the respective land trust or government agency.

2.3. Outcome datasets

2.3.1. Development

Conservation easements are the main tool for voluntary permanent land protection in the West, but their effectiveness is rarely the subject of rigorous quantitative assessment (Nolte et al., 2019). To test the assumption that private lands declared as conservation easements effectively restrict development in these ecologically sensitive and important mesic ecosystems, we made 30 m binary maps of development with developed and undeveloped classes using the 'Developed' class of annual Land Change Monitoring, Assessment, and Projection (LCMAP) Primary Land Cover (LCPRI) maps (Barber, 2022). We acknowledge that these maps likely underestimate low density exurban development, but there is no reason to expect these errors of omission are spatially biased, as others have noted with similar datasets derived from the Landsat time series (Nolte et al., 2019).

2.3.2. Mesic ecosystem quality

To test whether conservation easements had an effect on mesic vegetation we used the Water Resources Proportions (WRP) dataset produced from the Landsat time series within two Landsat path/row combinations (040029, 040028) in monthly Landsat composites from 2004 to 2020 (17 years) (Kolarik et al., 2024). We used mesic vegetation proportions in September, the last month of the water year, as this period marks the end of the long, dry summer in the High Divide when water availability is at its lowest (Silverman et al., 2019). We use mesic vegetation as a proxy for water availability during this time period, since it can only occur where it has access to water (either surface or subsurface) and surface water bodies remain largely undetected at the moderate spatial scales of publicly available satellite data (Kolarik et al., 2023). Since most mesic vegetation occurs in low-lying areas, we restricted our analysis to valley bottoms and areas of low slope using the USGS Landforms dataset (Theobald et al., 2015). We masked irrigated agriculture using the irrMapper dataset (Ketchum et al., 2020) as well as surface water in an attempt to focus solely on fully functional mesic ecosystems. For both analyses we used a stratified sample of 10,000 pixels to represent each tenure classification of interest to capture a wide range of variation while maintaining a manageable size for computation. If a pixel was ever encumbered with an easement at any time during the time series, we considered it an easement pixel rather than a non-easement. We used only private land pixels in this analysis.

2.4. Modeling

2.4.1. Quasi-experimental design

Evaluating the effectiveness of any conservation effort is difficult due to the absence of a traditional experimental design. When treatments are assigned (policies, restoration efforts, etc), there are no equivalent 'controls' to measure the outcome in the absence of treatment. To overcome the lack of explicitly identified control units, quasi-experimental impact evaluation methods can quantify the counterfactual outcome at the treatment site (Butsic et al., 2017). Quasi-

Table 1
Geospatial covariates used in matching and regression analyses. # indicates matching (easement status), $ indicates development, * indicates mesic vegetation.

Variable	Rationale	Reference	Source	Time varying
Elevation #$	High elevation plots are harder to access and less likely to be developed, thus more likely to be placed under easement.	(Graves et al., 2019)	National Elevation Dataset	no
Distance to public land #$	Locations closer to public land could improve connectivity and thus are more valuable to conserve	(Braza, 2017; Brown et al., 2022; Graves et al., 2019)	PADUS	no
Distance to towns #$	Land is more valuable closer to cities and thus more expensive and less likely to conserve	(Brown et al., 2022; Graves et al., 2019; Merenlender et al., 2004)	TIGER	no
Distance to land trust offices #	Locations closer to land trust offices are more likely to be placed under easement.	(Braza, 2017; Graves et al., 2019)	Land Trust Alliance	no
Distance to road #$	Locations farther from the road are more likely to be conserved (see distance to towns)	(Braza, 2017; Graves et al., 2019)	TIGER	no
Distance to GAP 1 or 2 #$	Locations closer to federally managed GAP 1 and 2 status are more likely to be targeted for easement declaration.	(Albers et al., 2008; Graves et al., 2019)	PADUS	no
County Income 2009 #$	Higher income counties are less likely to place locations under easement and more likely to develop	(Williamson et al., 2021)	US Census	no
County population change 2009–2020#$	Counties with higher population change are more likely to develop rather than conserve.	(Williamson et al., 2021)	US Census	no
Standardized Precipitation Evapotranspiration Index *	Climatic conditions affect water availability and vegetation conditions.	(Abatzoglou, 2013)	GRIDMET	yes
Slope *	Development and mesic ecosystems occur largely in flatter terrain.	(Ahmed and Jackson-Smith, 2019; Braza, 2017)	National Elevation Dataset	no
Contributing area *	Pixels with a larger contributing area are more likely to be wetter.	(Kolarik et al., 2023)	National Elevation Dataset	no
Value #$	More expensive locations are less likely to be placed under easement.	(Nolte, 2020)	PLACES	no

experimental designs account for non-random treatment assignments in observational studies that are often dependent on social and ecological factors which may also affect the outcome (Joppa and Pfaff, 2009; Reid et al., 2018). Statistical approaches that account for these considerations have proven to not only be effective ways to measure the effects of conservation interventions, but also necessary for reducing the impacts of selection bias associated with treatment assignment (Brandt et al., 2019; Simler-Williamson and Germino, 2022).

Researchers are increasingly relying on counterfactual statistical techniques to identify causal impact in conservation and ecological restoration settings in the absence of a traditional experimental design (Roopsind et al., 2019; Simler-Williamson and Germino, 2022; Sims et al., 2019). These techniques aim to reduce the likelihood of spurious inferences about a given treatment effect that may occur due to lack of consideration of selection bias associated with treatment assignment (Jones and Lewis, 2015). Matching is a way for researchers to reduce confounding biases in observational datasets where a traditional experimental design is unavailable (Butsic et al., 2017). For this analysis, we ultimately chose to explore nearest neighbor matching with replacement using Mahalanobis distance to match easement pixels on private land with non-easement private land pixels. A relatively simple way to measure similarity among observations, Mahalanobis distance matching relies on how close observations are to one another in euclidean space. Alternatively, more complex matching algorithms, like genetic matching, maximize balance among confounding variables in the dataset utilizing a nonparametric evolutionary algorithm (Diamond and Sekhon, 2006). We compared the outcomes with the more computationally intensive genetic matching algorithm, but found minimal differences in sampled pixels and balance despite higher computational cost. We used the R package 'MatchIt' and tested the sensitivity of the matched pixels samples to increasingly stringent caliper values measured in standard deviations relative to the distribution of a given covariate's values (Ho et al., 2011). Matched datasets are determined by how much the user allows the considered covariates to vary, as determined by the caliper measured in standard deviations. We settled on using a relatively stringent caliper size of 0.4, as this produced a reasonable sample size while also improving the balance in the dataset. We then used panel regressions in a Difference in Differences (DiD) approach that employs an interaction between samples that have been assigned a treatment and the period in which the treatment is assigned to estimate the treatment effect. A main assumption of DiD regressions is the assumption of parallel trends, which means both the treatment and control groups exhibit similar trajectories prior to treatment (Ham and Miratrix, 2023). We qualitatively assessed this assumption by plotting the outcomes over time in both groups and found reasonable agreement between them (Appendix S1).

2.4.2. *Covariate selection*

2.4.2.1. Matching. The main goal of matching in observational studies is to produce a balanced dataset of observations that were assigned treatment and those that were not relative to the observable variables that influence receiving the treatment (Jones and Lewis, 2015; King et al., 2011). We selected variables for matching based on their influence on easement likelihood as indicated by the literature (Table 1, indicated by #). Attributes of a given pixel associated with high likelihood of easement implementation often are what make that location undesirable for development. For instance sites at relatively high elevations (Graves et al., 2019) and farther from roads and towns (Brown et al., 2005; Merenlender et al., 2004) have higher likelihood of easement implementation. Conversely, for locations in counties with relatively high median income and population change, we expect a lower likelihood of easement implementation (Williamson et al., 2021). When a property is closer to a land trust office (Braza, 2017; Graves et al., 2019), or a protected area with GAP 1 or 2 conservation status (Albers et al., 2008; Graves et al., 2019), the likelihood of easement implementation increases. Lastly, pixels on properties that cost more are less likely to be placed under easement (Nolte, 2020). We collected or created relevant geospatial layers for each of these covariates to estimate each pixel's likelihood of easement implementation.

2.4.2.2. Regression. For regression analyses we developed a suite of covariates known to affect mesic vegetation occurrence and development, respectively (Table 1, indicated by * and $). For mesic vegetation, we used slope and contributing area as derived from a digital elevation model (Halabisky et al., 2023; Hird et al., 2017). We also used a one year moving average of the Standardized Precipitation Evapotranspiration Index (SPEI) to account for the influence of spatiotemporal temperature and precipitation variability on mesic vegetation abundance (Abatzoglou, 2013). In doing so we capture the full range of variation in temperature and precipitation throughout the year that might influence vegetation conditions.

For the development analyses, we collated covariates that indicate either land suitable or desirable for development. For example, elevation and slope determine a site's suitability for development, as sites at high relative elevations are harder to access and build on and sites on hillslopes also require more planning and engineering to develop. We control for county level population change from 2009 to 2020 and median income in 2009, with the rationale that wealthier counties and those experiencing the in-migration are more aligned with conservation action and thus less likely to be developed further (Williamson et al., 2018). We use 'distance to' metrics for cities and roads to control for accessibility differences, as pixels closer to city centers and with road access are more desirable and thus more likely to be developed. On the other hand, we use distance to public land and distance to land with GAP 1 or 2 conservation status to control for decreases in development likelihood as these distances get larger.

2.4.3. *Model specification*

We used Generalized Linear Mixed effects Models (GLMMs) to analyze our datasets. We standardized all continuous predictors to a mean of zero and unit variance. We fit all models using the *brms* package, a R-based wrapper to Stan, a probabilistic programming language that relies on Hamiltionian Monte Carlo Sampling to generate posterior distributions of the parameters of interest (Bürkner, 2017). We drew posterior samples for each model using four chains with 8000 iterations, with the first 4000 discarded as warmup. In all models, we used a random intercept term to account for spatial non-independence at the watershed level using HUC12 units. We chose these units due to the frequency at which we noticed management and collaborative groups (i.e. watershed councils) are organized at this level. We also used a random intercept to account for non-independence within each growing season, by treating each year as a factor. We used weakly informative priors (*Normal*(0,1)) for all regression models (Lemoine, 2019). With pixels as the unit of analysis, we used random subsets (20 %) of our sample pool (170,000 pixels for each land tenure designation) which led to representative, more computationally efficient samples.

The models we specified to represent the development process used a Bernoulli distribution with a logit link function, as development in each pixel had a binary (0,1) outcome rather than a proportion (Eq. (1a)). For all models using mesic vegetation proportion as the outcome variable in any given pixel, we used a Beta distribution that constrains modeled outcomes between 0 and 1 and a logit link function (Eq. (2a)). In all model structures, y represents individual observations (i) of either development or mesic vegetation proportion, which are nested within HUC12 units (j), and years (k). u represents the expectation of y, α represents intercepts (with varying intercept components for α_j and α_k for HUC12 and year which are normally distributed with standard deviations of σ), and $\mathbf{X}$ represents a matrix of covariates (Table 1). We evaluated the variance explained of each of our models using Bayesian R^2 (Gelman et al., 2019), and assessed model convergence using trace plots, by R-hat (< 1.05), and effective sample sizes in the bulk and tails of the sampling distributions (>1000) provided by *brms*.

$$y_{ijk} \sim Bernoulli\left(p_{ijk}\right) \tag{1a}$$

$$logit\left(p_{ijk}\right) = \alpha_{jk} + \tau\left(status_{ijk}\right) + \gamma\left(easement_{ijk}\right) + \omega\left(status_{ijk} * easement_{ijk}\right) + \boldsymbol{\beta}(\boldsymbol{X}) + \varepsilon_{ijk} \tag{1b}$$

$$\alpha_{jk} \sim Normal\left(\alpha_{jk}, \sigma_j\right) \tag{1c}$$

$$\alpha_k \sim Normal(\alpha, \sigma_k) \tag{1d}$$

$$y_{ijk} \sim Beta\left(u_{ijk}, \phi\right) \tag{2a}$$

$$logit\left(u_{ijk}\right) = \alpha_{jk} + \tau\left(status_{ijk}\right) + \gamma\left(easement_{ijk}\right) + \omega\left(status_{ijk} * easement_{ijk}\right) + \boldsymbol{\beta}(\boldsymbol{X}) + \varepsilon_{ijk} \tag{2b}$$

$$\alpha_{jk} \sim Normal\left(\alpha_{jk}, \sigma_j\right) \tag{2c}$$

$$\alpha_k \sim Normal(\alpha, \sigma_k) \tag{2d}$$

$$\phi \sim Gamma(0.1, 0.1) \tag{2e}$$

To demonstrate the usefulness of quasi-experimental design in observational studies like this one for reducing the effects of unobserved biases, we specified identical models to those described above, although without the DiD term (Eqs. (3a), (4a)). Instead, we used only dummy variables to describe the status of each pixel as either under easement or not, and include these in the regression models. We refer to these as 'naive' models hereafter.

$$y_{ijk} \sim Bernoulli\left(p_{ijk}\right) \tag{3a}$$

$$logit\left(p_{ijk}\right) = \alpha_{jk} + \tau\left(status_{ijk}\right) + \boldsymbol{\beta}(\boldsymbol{X}) + \varepsilon_{ijk} \tag{3b}$$

$$\alpha_{jk} \sim Normal\left(\alpha_{jk}, \sigma_j\right) \tag{3c}$$

$$\alpha_k \sim Normal(\alpha, \sigma_k) \tag{3d}$$

$$y_{ijk} \sim Beta\left(u_{ijk}, \phi\right) \tag{4a}$$

$$logit\left(u_{ijk}\right) = \alpha_{jk} + \tau\left(status_{ijk}\right) + \boldsymbol{\beta}(\boldsymbol{X}) + \varepsilon_{ijk} \tag{4b}$$

$$\alpha_{jk} \sim Normal\left(\alpha_{jk}, \sigma_j\right) \tag{4c}$$

$$\alpha_k \sim Normal(\alpha, \sigma_k) \tag{4d}$$

$$\phi \sim Gamma(0.1, 0.1) \tag{4e}$$

3. Results

We compiled descriptive statistics of total area, valley bottoms, and mesic vegetation associated with each land tenure category to quantify general patterns of mesic ecosystems as they pertain to land tenure (Table 2). Private lands contain a higher area of valley bottoms than public lands. Valley bottoms on private lands have a higher proportion

Table 2
Descriptive statistics for total area of public, private, and conservation easements, mesic vegetation area in the defined valley bottom area (excluding irrigated agriculture), and the interquartile range and median values of mesic vegetation proportion of valley bottom pixels for all land tenure distinctions in the study area.

	Area (km2)			Mesic vegetation proportion (%)		
	Total Area	Valley Bottom	Mesic vegetation	25th percentile	Median	75th percentile
Private (all)	18,395	4482	1803	4.28	20.7	48.83
Easements	1254	272	161	18.94	42.38	63.47
Public	51,059	8047	1530	2.71	9.76	23.05

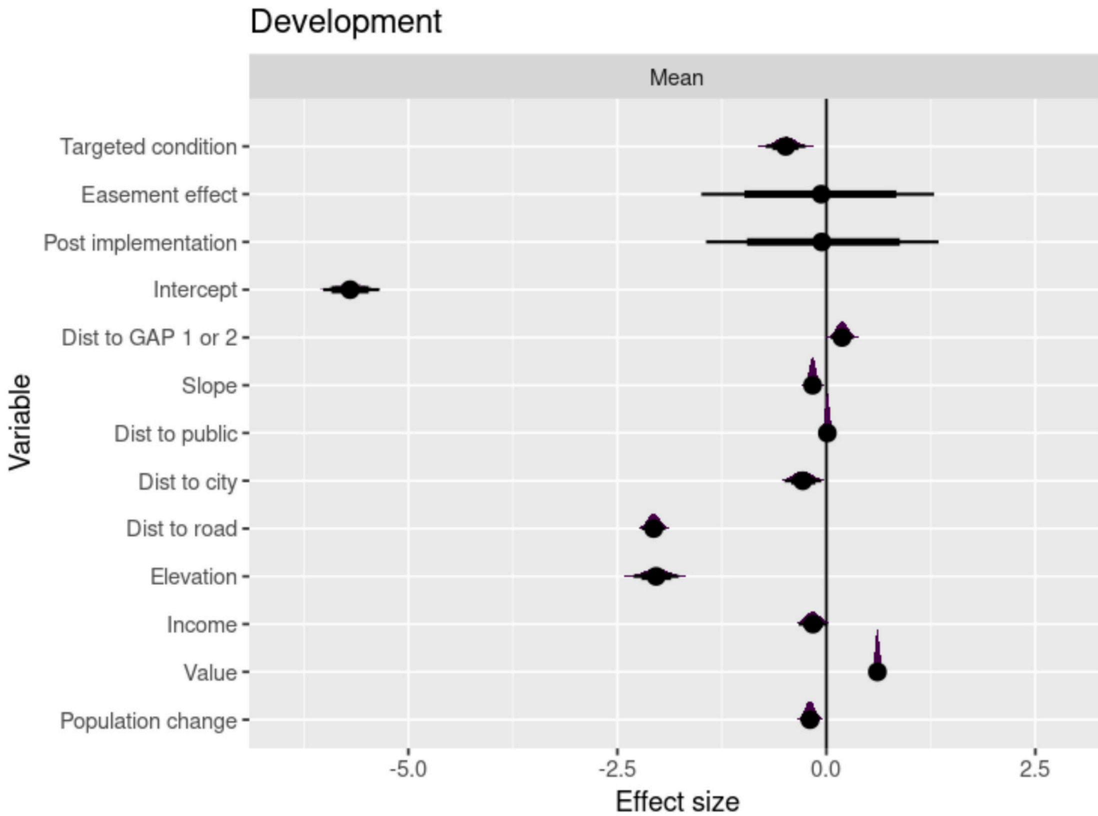

Fig. 3. Effect sizes of targeted areas (Targeted condition) and impacts of easements on development (Easement effect) and other predictors in the binomial regression model specified to measure differences in development on conservation easements versus undeclared private land. Points represent the mean parameter estimates, thick bars represent 80 % credibility intervals (CIs), and thin bars represent 95 % CIs.

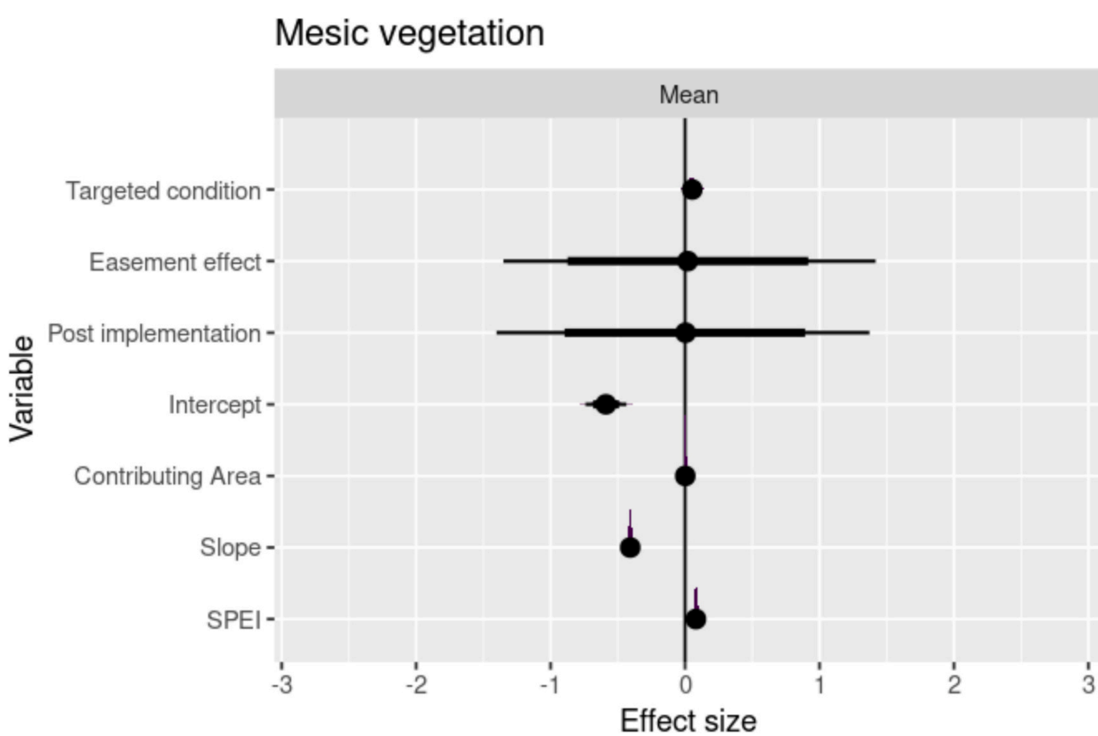

Fig. 4. Effect sizes of targeted areas (Targeted condition) and impacts of easements on mesic vegetation (Easement effect), predictors in the beta regression model specified to measure differences in mesic ecosystem quality on conservation easements versus undeclared private land using the matched panel. Points represent the mean parameter estimates, thick bars represent 80 % CIs, and thin bars represent 95 % CIs.

of mesic vegetation than valley bottoms on public lands. Valley bottoms of easements have a higher proportion of mesic vegetation than those found on all private lands. Although public lands are nearly three times more prevalent than private lands, more than half of the mesic vegetation in the valley bottom areas is found on private lands, even with irrigated agriculture excluded. Further, the spread of the interquartile range (IQR) of mesic vegetation percentage in valley bottom pixels on private (4.28 % to 48.83 %) versus that on public lands (2.71 % to 23.05 %) is substantially higher. Together, these metrics confirm that private lands typically hold higher quality mesic ecosystems that are essential to landscape conservation efforts. When we isolate easements specifically, we show that the IQR of mesic vegetation proportion (18.94 % to 63.47 %) is higher than in valley bottoms on all private lands including non-easements, indicating higher quality habitats found on easements.

In the sample pool, 464 of 496 easements have known easement holders (Table S1). 22 different easement holders were responsible for the remaining 464 easements. We closely reviewed the mission statements of these 22 easement holders individually for keywords that would communicate a commitment to biodiversity conservation that mesic habitats would be central to their success: *wildlife, habitat, wetland (s), river(s), riparian, ecosystem(s)*. 14 of the easement holders mentioned at least one of these keywords, 3 had a mission statement, but did not mention any of these themes, and 5 did not have a mission statement easily accessible. The three easement holders with mission statements that did not use these words were The Nature Conservancy (TNC), the Bureau of Land Management (BLM), and the U.S. Forest Service (USFS).

In total, there are 310,661 acres under easement in the study area, with 58,664 (∼19 %) designated as GAP 1 or 2, the designation statuses managed for biodiversity (Table S2). Despite not having biodiversity conservation keywords, TNC holds >90 % of easement acreage designated as either GAP 1 or 2. The BLM holds three easements totaling 81 acres, 75 of which belong to two easements with a GAP 1 status, and the USFS 3455 acres of easements with GAP 2 status. The easement holder responsible for the most acreage is the Montana Land Reliance with 70,201 acres total (39,904 acres of GAP 3, 30,297 acres of GAP 4). Of the 310,661 total acres under easement, only 44,047 were easements held by organizations that did not include conservation keywords in their mission statements or did not have GAP 1 or 2 conservation status, indicating that over 85 % of easements were either held by organizations focused on conservation of biodiversity, habitat, ecosystems, or riparian systems or intended to improve them.

3.1. Do easements effectively target high-quality lands (i.e. low development, high mesic) for protection?

Our results demonstrate that easements targeted higher quality mesic ecosystems compared to non-easements. Prior to implementation, easements were ∼ 39 % (95 % Credibility Interval (CI) 52 % to 22 %) less likely to have been developed than non-easements (Fig. 3). In terms of mesic ecosystems, prior to implementation, easements had marginally higher proportions of mesic vegetation than non-easements (1.22 %, 95 % CI: −0.39 % to 2.89 %. 80 % CI: 0.16 % to 2.31 %) (Fig. 4).

3.2. Do easements lead to high-quality conditions once they are implemented?

We do not find evidence that easements led to less development compared to non-easements. The effect of easement implementation on development is weakly negative, estimating pixels declared as easement are ∼6 % less likely to become developed on average. However, this estimate is highly uncertain and variable, as the 95 % CI ranges from ∼76 % less likely to 278 % more likely to be developed (Fig. 3). Similarly, we found no distinguishable effect of easement implementation on mesic vegetation proportion (Fig. 4). The mean estimate of this effect falls almost exactly on zero and is highly uncertain (95 % credibility interval (CI): −1.46 - 1.36). This translates to a negligible estimated difference in mesic vegetation proportion between declared easement pixels and those that are not (∼0 % difference in mesic vegetation proportion, 95 % CI: −23.33 % to 33.59 %). For more details regarding other model parameters, please see Appendix S1.

3.3. Comparisons with naive model specifications

We found that when we use a naive (i.e. non-counterfactual

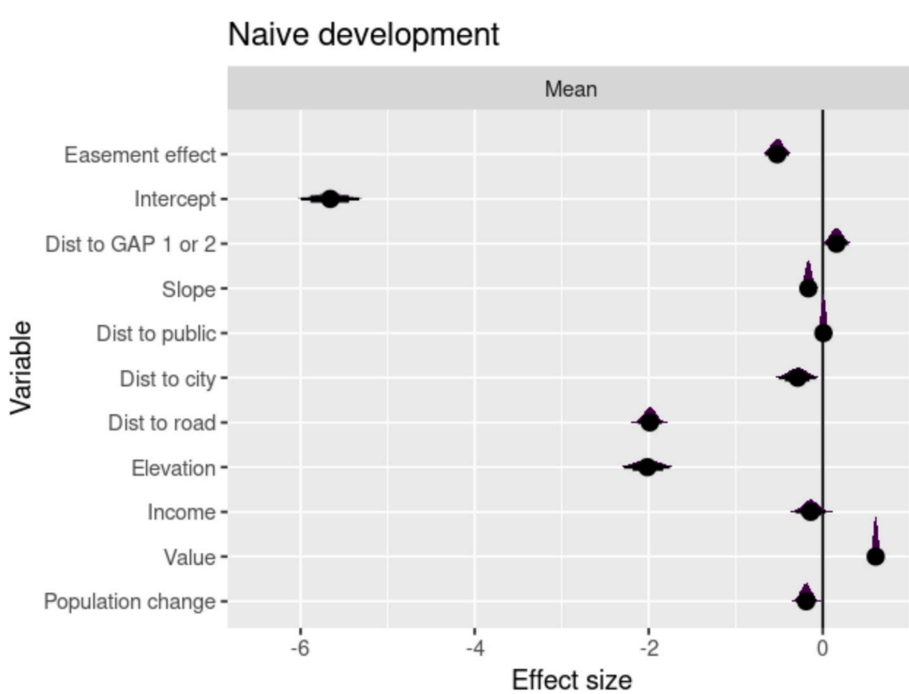

Fig. 5. Effect sizes of predictors in the naive regression model for development. Points represent the mean parameter estimates, thick bars represent 80 % CIs, and thin bars represent 95 % CIs.

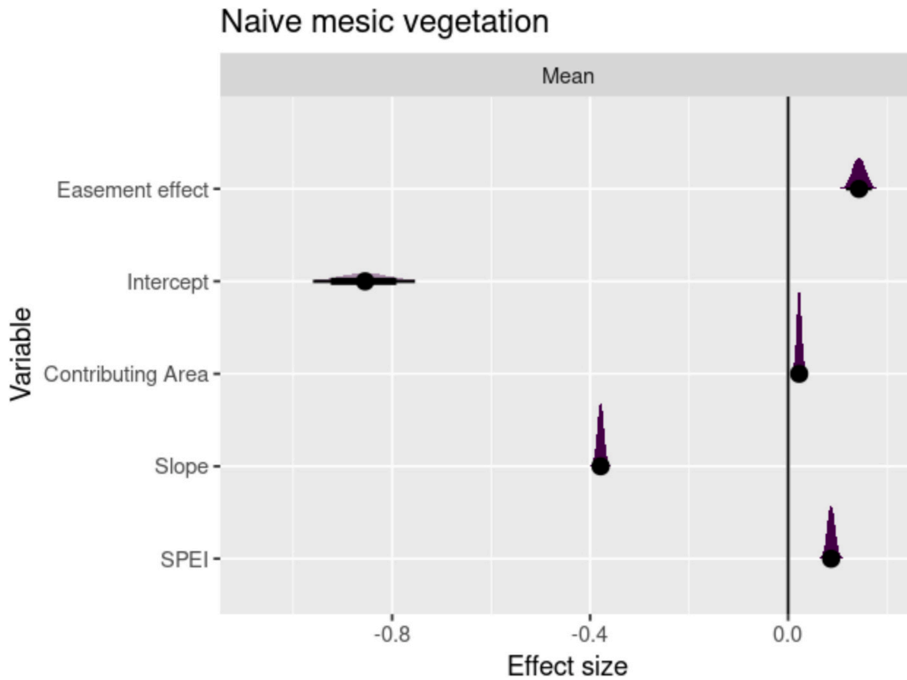

Fig. 6. Effect sizes of predictors in the naive regression model for mesic ecosystem quality. Points represent the mean parameter estimates, thick bars represent 80 % CIs, and thin bars represent 95 % CIs.

approach), our interpretation of the influence of this designation on both development and mesic vegetation changed. With the naive analysis, we found that easements result in a 40.86 % (95 % CI: 48.42 % to 32.14 %) decrease in development likelihood (Fig. 5) and a 3.09 % (95 % CI: 2.51 % to 3.67 %) increase in proportion of mesic vegetation (Fig. 6).

3.4. Model fit

Median Bayesian R^2 values ranged from 0.33 to 0.361 in the models we specified with consistently low uncertainty for all. We estimated the variance explained by the development DiD regression as $R^2 = 0.361$ (95 % CI: 0.351 to 0.371). The variance explained by the mesic ecosystem on private lands DiD regression with pre-matching is $R^2 = 0.335$ (95 % CI: 0.328 to 0.341). In the naive model specifications, we estimated variance explained as $R^2 = 0.33$ (95 % CI: 0.318 to 0.341) and $R^2 = 0.33$ (95 % CI: 0.325 to 0.334) for development and mesic ecosystem models respectively.

4. Discussion

We found that easements, on average, are placed on lands with less development and higher quality mesic ecosystems compared to non-easements. This result indicates that land trusts and other organizations working to conserve ecologically meaningful swaths of land are in fact identifying high quality areas for conservation. Thus, easements increased the potential for effective threatened species conservation because previously these habitats were absent from protected area networks (Pimm et al., 2014). This finding contributes to the discussion about whether easements are effective tools for protecting habitats for federally listed endangered species and providing connectivity among ecologically intact regions (Brown et al., 2022; Graves et al., 2019; Merenlender et al., 2004; Rissman et al., 2007). We consider our results to be a positive review for easement holders working to place lands with high ecological value under easement, ideally conserving them in perpetuity.

We did not find strong evidence that easements lead to improved outcomes following implementation. Instead, we found that on average, easements do not limit development or improve mesic ecosystem condition compared to non-easements. Furthermore, our results indicate that easements can result in a mix of outcomes, as indicated by the wide credibility intervals of our model estimates. This uncertainty could also be in part due to the relatively coarse nature of the outcome datasets, particularly with respect to development not detectable in 30 m pixels. In short, our results indicate that some easements restrict development and/or increase the proportion of mesic vegetation, and others do not. The outcomes are no different than those in private pixels not protected with a conservation easement. This finding is likely a result of the ways in which easements specifically, and private land protection generally, are negotiated, implemented, and regulated. Because easements are negotiated with an individual landowner, what is allowed and not allowed in terms of development and habitat conservation varies for each easement. For example, with respect to development, landowners may negotiate to be permitted to construct previously planned development for which they have not yet broken ground (Rissman, 2013). Accordingly, it is reasonable to expect variation in outcomes (Rissman et al., 2007). In terms of monitoring and enforcement, the specifics of each easement agreement are time consuming to obtain, and determining whether structures and roads were constructed illegally would require continued oversight, monitoring, and enforcement by the organization that holds the easement (Morris, 2008).

Our results indicate that the current way in which easement programs are evaluated, by focusing largely on acres conserved, may be leading to a missed opportunity for conservation. Many argue that restoring, above and beyond simply protecting, keystone ecosystems is necessary for conservation in the anthropocene (Biermann, 2012). Currently, easement holders are not required to focus on improving and/or maintaining the ecological integrity of the lands they placed under easement (Brown et al., 2022). Organizations that hold easements often acquire the right to perform restoration activities on the property, but have no obligation to do so (Rissman, 2013). The variability in outcomes that we observe could be reduced by encouraging, if not expecting, landowners to improve easement properties for biodiversity and habitat. Even minimal conservation and restoration goals that reflect the mission statements or the majority of easement holders could potentially help to reduce the variability in outcomes if explicitly requested. Otherwise, as has been documented elsewhere, easement enrollment does not necessarily lead to improved ecological conditions (Byrd et al., 2009), and could be considered a sub-optimal use of limited conservation dollars.

Another way in which easement programs could improve their ecological potential is to overcome challenges associated with monitoring outcomes on private lands. Monitoring outcomes relative to conservation and restoration goals is integral to measure, communicate, and adaptively manage projects. We show that geospatial datasets provide low cost opportunities for monitoring, and this is in line with much of the discussion regarding monitoring as a path towards improving the efficacy of easements for conservation rather than preservation (Braza, 2017; Kiesecker et al., 2007; Merenlender et al., 2004; Rissman et al., 2015). While in situ monitoring is often expensive and time consuming, we show that maps derived from freely available remotely sensed imagery can be used at least for high level monitoring tasks (Malakoff and Nolte, 2023; Tsalyuk et al., 2015). We are not suggesting that property visits do not have inherent value; rather, we suggest that augmenting these visits with geospatial analyses using freely available earth observation data can guide discussions for effective adaptive management of easements and compliance with the respective negotiated agreements (Wiens et al., 2009). Freely available data are useful for high level monitoring tasks, but detailed site information should be collected in situ. We acknowledge that no panacea exists for private land conservation but the use of easements as a conservation tool has been referred to as “a car with no one in the driver's seat” (Morris, 2008), and we posit that geospatial datasets can help steer these tools towards positive social-ecological outcomes. Evidence from aggregate analyses like this one can reveal impacts and changes brought about by various conservation investments.

In terms of how geospatial data can be used in a counterfactual framework to identify causal impacts of easement effectiveness, our study generated important insights. First, we found that a counterfactual approach is necessary to accurately measure the easement impact. Using a non-counterfactual approach (i.e. a regression analysis with the same panel data that does not include the interaction between a pixel declared as an easement and the implementation period) led to an erroneous interpretation of the effect of the easement. We posit that using the DiD approach with panel data as we have done here is necessary for disentangling the effects of land management from the choices that led to a given management decision. Our results reflect the idea that observable and unobservable forms of bias that lead to management decisions must be incorporated into causal impact analyses for them to be effective (Butsic et al., 2017; Jones and Lewis, 2015). Without statistical approaches that account for these considerations, spurious inferences about the efficacy of a given conservation outcome are likely. Second, we ultimately decided that the use of pre-matching in our study area was helpful, but maybe not necessary, as results did not change drastically when we omitted that step. This is not an entirely surprising result given our sampling design and the constraints we imposed on our study area. Not only are two Landsat footprints a rather small area to expect drastic differences in terms of drivers of easement implementation, but we also focused our analyses to valley bottoms only, decreasing many of the possible sources of significant variation that could lead to bias in our models (Butsic et al., 2017).

In this study, we focused solely on the effects of easements without analyzing other property purchases by land trusts such as fee simple acquisitions. We acknowledge that these types of property acquisitions are more likely to result in beneficial environmental outcomes as the conservation actors that purchase these have more control over the activities that occur on the land despite return on investment and prioritization mismatches highlighted recently due to higher costs (Le Bouille et al., 2023). We also acknowledge that focusing on mesic vegetation as the outcome does not directly measure biodiversity, habitat quality, or landscape connectivity, but is only a proxy for these phenomena. Until datasets with sufficient spatial and temporal resolution for these explicit outcomes of interest are made available, we contend that mesic vegetation estimates can provide a high-level context for these. We excluded irrigated agriculture from our analyses to focus on ecologically intact areas of mesic vegetation. The use of irrigated agriculture by species of interest in our study area has been well documented (Barker et al., 2019; Donnelly et al., 2016, 2024), but the extent of its functions are limited and adverse effects are also well discussed (Sterling et al., 2013), leading to our decision to exclude them from the analysis. Further, we did not investigate ‘leakage’ effects (i.e. negative spillovers) as is often done in analyses of environmental policies and protection (Roopsind et al., 2019). In this case we do not expect that protection of mesic ecosystems in some places would lead to degraded mesic ecosystems in others. In fact, it would be reasonable to expect positive rather than negative spillovers in neighboring areas due to connectedness of healthy riparian systems (Pollock et al., 2014). We also did not explicitly model errors associated with spatial autocorrelation, but used a random sample of pixels to capture the distribution of errors that may exist across the study area. Future research directions could involve the integration of publicly available water rights records and how they relate to easement incentives, as this link could be integral in understanding outcomes for riverscapes. Another possible research avenue could focus on whether the landowners receiving tax benefits from easements are in the agricultural or development sector, as some cases show that developers will buy land, develop a portion of it, and then implement easements on a small fraction to reap the tax benefits (Stephens and Ottaway, 2003). The evaluation of the specifics of easements are likely to be very difficult, however, because of the lack of information available to the public, as easements are not required to be included in the National Conservation Easement Database (Rissman et al., 2017; Williamson et al., 2021).

5. Conclusion

Conservation easements are an integral part of the conservation landscape but their efficacy has been questioned in recent years. We provide evidence that the organizations placing land under easement are effectively identifying ecologically valuable areas of land to conserve. However, whether easements effectively enhance ecosystem function or limit development varies widely. With easements being such a popular method of private land conservation, we posit that largely, these are missed opportunities for effective conservation of ecosystems that have disproportionate ecological importance and that are also largely absent from the public protected area network. Including explicit restoration and conservation expectations that align with the mission statements of easement holders into easement agreements could help easements achieve their potential as a biodiversity conservation tool. While many before us have called for more rigorous monitoring of easements and their outcomes, we provide examples of how freely available earth observation data can help to monitor high level impacts of easements on mesic habitats that are critical for maintaining biodiversity and landscape connectivity in dryland regions. These assessments using geospatial data should operate in tandem with in situ assessments of easement compliance, because together they can provide insights for adaptive management of easements and other private land conservation tools.

Github

GEE repo

CRediT authorship contribution statement

Nicholas E. Kolarik: Writing – review & editing, Writing – original draft, Visualization, Software, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Megan Cattau:** Writing – review & editing, Methodology. **Carolyn Koehn:** Writing – review & editing, Methodology, Investigation. **Anand Roopsind:** Writing – review & editing, Methodology, Formal analysis.

Matthew Williamson: Writing – review & editing, Software, Methodology, Formal analysis. **Jodi Brandt:** Writing – review & editing, Writing – original draft, Supervision, Resources, Methodology, Investigation, Funding acquisition, Formal analysis.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.biocon.2025.111234.

Data availability

Data will be made available on request.

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