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Effectiveness of protected areas in the Caucasus Mountains in preventing rangeland degradation

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Abstract
As land use intensifies globally, it increasingly exerts pressure on protected areas. Despite open, nonforested landscapes comprising up to 40% of protected areas globally, assessments have predominately focused on forests, overlooking the major pressures on rangelands from livestock overgrazing and land conversion. Across the southern Caucasus, a biodiversity hotspot extending over 5 countries, we conducted a broadscale assessment of the extent to which protected areas mitigate land-use pressure on rangelands in them. Using satellite-based indicators of rangeland vegetation greenness from 1988 to 2019, we assessed the effectiveness of 52 protected areas. This period encompassed the collapse of the Soviet Union, economic crises, armed conflicts, and a major expansion of the protected area network. We applied matching statistics combined with fixed-effects panel regressions to quantify the effectiveness of protected areas in curbing degradation as indicated by green vegetation loss. Protected areas were, overall, largely ineffective. Green vegetation loss was higher inside than outside protected areas in most countries, except for Georgia and Turkey. Multiple-use protected areas (IUCN categories IV–VI) were even more ineffective in reducing vegetation loss than strictly protected areas (I & II), highlighting the need for better aligning conservation and development targets in these areas. Mapping >10,000 livestock corrals from satellite images showed that protected areas with a relatively high density of livestock corrals had markedly high green vegetation loss. Ineffectiveness appeared driven by livestock overgrazing. Our key finding was that protected areas did not curb rangeland degradation in the Caucasus. This situation is likely emblematic of many

regions worldwide, which highlights the need to incorporate degradation and nonforest ecosystems into effectiveness assessments.

KEYWORDS
biodiversity hotspots, grasslands, grazing pressure, impact evaluation, land degradation, livestock grazing, matching statistics, steppes

INTRODUCTION

Protected areas are a cornerstone of conservation efforts worldwide, established to safeguard species and habitats and to maintain ecosystem integrity and services (Bruner et al., 2001; Watson et al., 2014). There has been a marked expansion of the protected area network in the past decades, partially driven by Aichi target 11 (Maxwell et al., 2020), and this trend is poised to continue with the recently agreed-on target of protecting 30% of Earth's land surface by 2030 (CBD, 2022). Although more land is protected, human pressures on these lands increase (Jones et al., 2018). That is why it is important to understand the effectiveness of protected areas in curbing human pressures and delivering biodiversity targets (Jones et al., 2018; Laurance et al., 2012).

Land use affects protected areas in various ways (Jones et al., 2018), through agricultural activities, forestry, urban area development, extension of transportation network, and expansion of energy production (Schulze et al., 2018). These pressures have manifold impacts, including, but not limited to, habitat loss, fragmentation, and degradation (Banks-Leite et al., 2020). However, some land use (e.g., livestock grazing, mowing) can be expected or even desired inside protected areas, particularly in multiple-use areas or where the history of human habitation is long (Mammides, 2020). Additionally, certain land management practices may align well with protected areas' goals, such as controlled fire or mowing to maintain seminatural grasslands (Gavin et al., 2018). Accordingly, a better understanding of the baseline land-use practices in protected areas and their surroundings and how these have changed over time is needed for assessments of protected area effectiveness (Pressey et al., 2015, 2021).

The effectiveness of protected areas varies widely (Jones et al., 2018; Laurance et al., 2012). Encouragingly, protected areas are generally successful in protecting forests (Geldmann et al., 2013), and even if they do not prevent it completely, forest loss is reduced (Wolf et al., 2021), albeit with differences in effectiveness among countries and over time (Butsic, Munteanu, et al., 2017). Although much is known about protected area effectiveness in stemming forest loss, which is relatively easy to measure from remotely sensed data (Ghoddousi et al., 2022), about 40% of the world's protected areas are not forested (i.e., have <20% tree cover), including steppes, rangelands, deserts, tundra, and alpine areas (European Commission, 2018). The effectiveness of these protected areas in shielding them from land-use impacts is largely unknown (Bai et al., 2008; Song et al., 2018). It is thus necessary to expand assessments of protected area effectiveness beyond forested landscapes.

One challenge in this is that land use in nonforested ecosystems typically does not lead to the full conversion of land cover but rather modifies natural vegetation gradually and in more subtle ways, which is difficult to measure (Dubovyk, 2017). For example, livestock grazing, a common human land use inside protected areas, may cause widespread habitat degradation but is much more challenging to define and detect than forest loss (Schleicher et al., 2017; Soofi et al., 2018). Degradation is assumed to be particularly widespread in the world's nonforested ecosystems (ILRI et al., 2021; Strömberg & Staver, 2022). With the UN Decade on Restoration (Dudley et al., 2020) and target 2 of the Global Biodiversity Framework, reversing degradation is now a global priority, the goal of which is to effectively restore at least 30% of degraded ecosystems by 2030 (CBD, 2022). Tracking habitat degradation and recovery should thus be a basis for assessing the effectiveness of area-based conservation in such systems. However, this is challenging because defining habitat degradation necessitates an understanding of the social–ecological context of the drivers of degradation, and detecting degradation at scale is difficult because satellite-based indicators for degradation in nonforest ecosystems are not readily available (Dubovyk, 2017; Gibbs & Salmon, 2015; Vogt et al., 2011). Developing and testing approaches for assessing whether protected areas reduce degradation in nonforested ecosystems is thus a priority.

Our focus was on rangelands, by which we mean shrublands, grasslands, and steppes, covering 54% of the global terrestrial surface (ILRI et al., 2021). Rangelands are among the most widespread nonforest ecosystems, hosting rich biodiversity, storing large amounts of carbon, and providing food for millions of people (Strömberg & Staver, 2022). They are also particularly vulnerable to degradation due to livestock overgrazing and anthropogenic fires (Bond & Parr, 2010; Strömberg & Staver, 2022). Despite the importance of rangelands and the strong land-use pressures they are exposed to, the patterns of rangeland degradation in- and outside protected areas are largely unknown. Evaluations of protected area effectiveness focusing on rangelands are rare (but see Mammides et al. [2024]).

Recent advances in remote sensing provide an opportunity to better estimate rangeland degradation because long-term and frequent measurements of vegetation are now available (Gibbs & Salmon, 2015; Lewińska et al., 2020; Piipponen et al., 2022). Traditionally, rangeland status and degradation have been assessed using vegetation indices, such as the normalized difference vegetation index (NDVI), to approximate vegetation vitality and cover and, when tracked over time, to assess degradation (de Jong et al., 2011). However, vegetation indices can be inaccurate in sparse vegetation cover where bare soil is widespread, which is common in rangelands (Elmore et al., 2000; Huete et al., 1985; Smith et al., 2019). Spectral unmixing addresses this issue by estimating actual ground cover fractions (e.g., green vegetation, soil) (Hostert et al., 2003; Smith et al., 2019). Assessing these ground cover fractions over time can provide insights into the drivers of rangeland degradation, such as climate change and overgrazing (Stanimirova et al., 2019). With sufficient availability of satellite observations from new acquisitions and image archives and with increased computational capacities, spectral mixture analysis is now possible across large geographic extents and over longer periods (Frantz et al., 2022). Such indicators of rangeland vegetation greenness can serve as proxies for rangeland degradation, and integrating these indicators in the assessments of protected area effectiveness should therefore facilitate moving beyond the prevailing bias of effectiveness studies toward forested protected areas (Ghoddousi et al., 2022).

The Caucasus Ecoregion, between the Black and Caspian seas, is one of the most biologically diverse and culturally rich regions on Earth (Krever et al., 2001). Around 39% of the Caucasus is composed of rangelands (Bleyhl et al., 2017), which have been subject to seminomadic pastoralism and agriculture for millennia (de Leeuw et al., 2019; Neudert, 2021). However, in the 20th and 21st centuries, the Caucasus has experienced major land-use changes (Buchner et al., 2020; Neudert, 2021). For example, the collapse of the Soviet Union and multiple armed conflicts that followed in the 1990s caused major changes in institutions and livelihoods, while also intensifying exploitation of natural resources in parts of the region (Brandt, 1992; Radeloff et al., 2013). Likewise, there were major land-use changes in the region's rangelands due to widespread cropland abandonment (Baumann et al., 2014; Buchner et al., 2020, 2022) and steep fluctuations in livestock numbers (FAO, 2023) but also to violations of protected area policies, for example, via illegal livestock grazing (Krever et al., 2001). Concomitantly, there has been a major expansion of the protected area network, with a general shift in management practices from strict protection to multiple-use landscapes (Gunya et al., 2021; Montalvo Mancheno et al., 2016). Despite these major changes, the effectiveness of protected areas in safeguarding rangelands in the Caucasus is unclear (Bragina et al., 2015; Montalvo Mancheno et al., 2016). Additionally, the role of livestock, as a key pressure on protected areas in the region, is unknown (Neudert, 2021).

We focused on 52 protected areas in 5 countries in the southern Caucasus, where rangelands are the dominant land cover and play a key role in the livelihood of local communities (Buchner et al., 2020; Neudert, 2021). We analyzed fractional green vegetation derived from Landsat satellite image time series from 1988 to 2019 and applied a rigorous impact evaluation framework. Further, we assessed the correlation between livestock presence, as one of the major drivers of land-use change in this region, and rangeland green vegetation loss inside protected areas (Neudert, 2021). Specifically, we addressed the following questions: Have protected areas been effective in curbing rangeland degradation, as approximated through green vegetation loss? How does effectiveness differ among countries, management categories (strictly protected vs. multiple-use areas), and protected areas of different sizes and ages? And, what is the association between livestock presence and green vegetation loss inside protected areas?

METHODS

Study area

We focused on the southern Caucasus (Figure 1), which we defined according to the Caucasus Ecoregion boundary (Krever et al., 2001); the Kura River, Surami Pass, and Rioni River demarcated the northern boundary. The area covers all of Armenia and parts of Azerbaijan, Georgia, Iran, and Turkey. Most of the southern Caucasus consists of mountainous rangelands, with vegetation characterized by open juniper woodlands, steppes, mountain grasslands, and occasionally broadleaved forests (Bleyhl et al., 2019; Buchner et al., 2020). The rangelands in the Caucasus have high species richness, including plants (~2800 species, many endemic to rangelands), amphibians (e.g., Caucasian salamander [*Mertensiella caucasica*]), reptiles (e.g., Caucasian viper [*Vipera kaznakovi*]), and birds (e.g., Caucasian grouse [*Lyrurus mlokosiewiczi*]) (Zazanashvili et al., 2020). This region is also home to a wide diversity of threatened megafauna, such as bezoar goat (*Capra aegagrus*), brown bear (*Ursus arctos*), mouflon (*Ovis gmelini*), and Persian leopard (*Panthera pardus tulliana*), which have their core populations in protected areas.

We included a selection of protected areas in IUCN management categories I–VI from the World Database of Protected Areas (www.protectedplanet.net). We excluded protected areas representing lakes and wetlands ($n = 9$). We only included protected areas established before 2013, given the temporal extent of our vegetation greenness data (1988–2019; see below) and that the effects of conservation interventions may not appear in newly established protected areas ($n = 9$) (Bruner et al., 2001). Because the impact of protected areas may not always match their management objectives (Guan et al., 2021; Leberger et al., 2020; Leroux et al., 2010), we visually checked the land-cover maps of protected areas in IUCN management category III and removed them from the analyses ($n = 22$). These natural monuments represent caves, canyons, mountain peaks, or similar features with little to no rangeland cover in the southern Caucasus. Two protected areas were in the conflict zone between Armenia and Azerbaijan, so we assumed they were not operational during the study period and removed them. The remaining 52 protected areas (Appendix S1) were distributed among the countries as follows: Armenia 16, Azerbaijan 11, Georgia 8, Iran 9, and Turkey 8, and they had a wide range of IUCN management categories (I, II, IV, and V; none in category VI) and establishment dates (1929–2011) (Figure 1). We divided the protected areas according to management categories into strictly protected (i.e., IUCN category I & II) and multiple-use areas (i.e., IUCN categories IV & V) (Elleason et al., 2021). The total area of the selected protected areas was around 14,100 km^2.

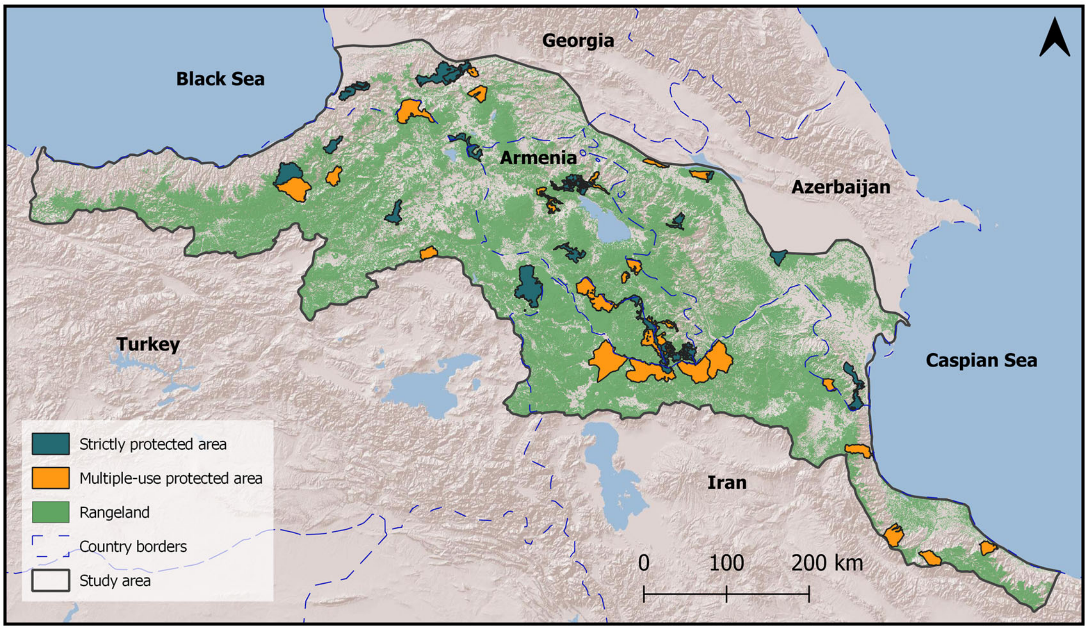

FIGURE 1 Study area in the southern Caucasus showing the distribution of rangelands (Bleyhl et al., 2017; Buchner et al., 2020) and the location of the selected strictly protected (IUCN categories I & II) and multiple-use areas (IV–VI) in Armenia, Azerbaijan, Georgia, Iran, and Turkey.

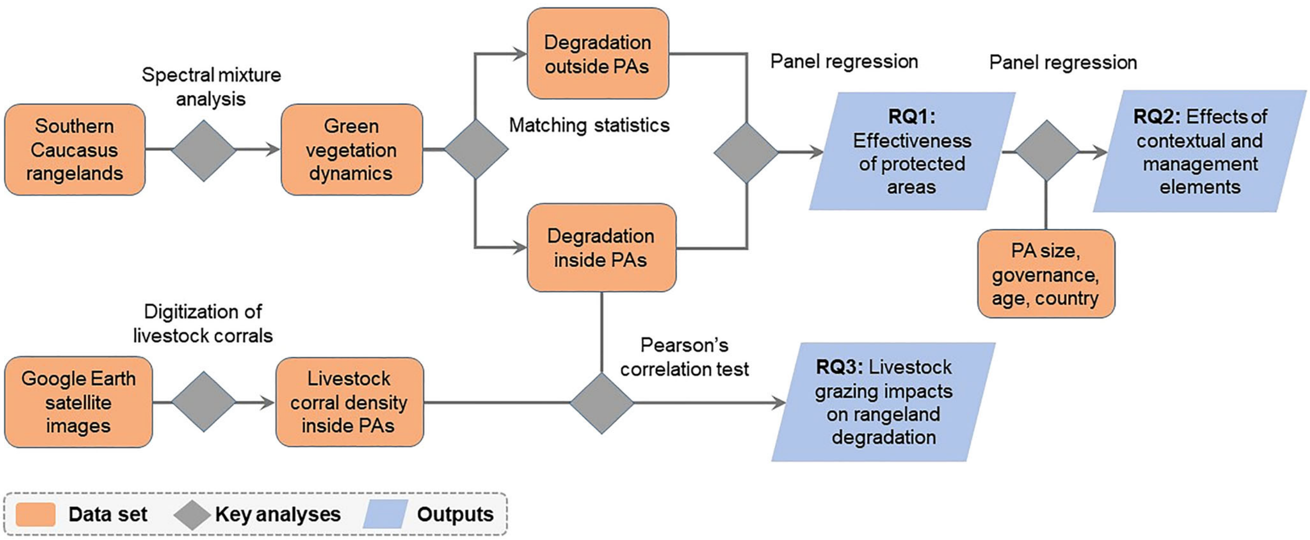

FIGURE 2 Analytical workflow showing how we assessed the effectiveness of protected areas (PAs) in preventing rangeland degradation (i.e., loss of green vegetation) in the southern Caucasus (RQ1), measured the impacts of contextual and management elements on effectiveness (RQ2), and quantified the role of livestock grazing on green vegetation loss in protected areas (RQ3).

Data analyses

We assessed the impact of protected areas on rangeland green vegetation loss in the southern Caucasus in 3 steps (Figure 2). First, we tested the effectiveness of protected areas in reducing rangeland green vegetation loss by comparing pixels inside and outside protected areas. Second, we determined whether contextual and management elements, such as country, IUCN category, size or the age of protected areas, influenced the level of effectiveness against rangeland green vegetation loss. Finally, we assessed whether livestock grazing, based on density of summer pasture corrals, determined the level of green vegetation loss inside the protected areas.

Degradation indicators

We used the land-cover maps by Buchner et al. (2020) and Bleyhl et al. (2017) to identify rangelands inside and outside protected areas across the southern Caucasus. To approximate degradation in these rangelands, we analyzed time series of green vegetation cumulative endmember fraction (CEF), which represents annual sums of green vegetation ground cover fractions measured in our case every month (Lewińska et al., 2020). The time series derived by Lewińska et al. (2021) was based on about 43,800 unmixed Landsat satellite images acquired from 1988 to 2019 and decomposed to 4 fractional covers: soil or bare rock, green vegetation, dry vegetation, and shade. Because ground cover fractions change over the course of a year due to changes in phenology and illumination (Kuemmerle et al., 2006), summing fractional cover derived at constant intervals over a calendar year normalizes these changes. To assess green vegetation loss at 30-m resolution, we analyzed changes in green vegetation CEF represented as a percentage point change and scaled from 0 to 1 (Lewińska et al., 2020). A decrease in green vegetation fraction over time can either result in a change from green vegetation to soil (i.e., green vegetation loss) or dry vegetation (i.e., desiccation) (Lewińska et al., 2020). We masked pixels with missing observations for a given year. More details on the estimation of our rangeland green vegetation loss measure are in Lewińska et al. (2020, 2021).

Impact evaluation

We selected a random sample of 347,632 rangeland pixels, 53,816 inside and 293,816 outside protected areas, with a minimum distance of 500 m to limit the effects of spatial autocorrelation for further analyses. To account for potential leakage and blockage effects from protected areas (Fuller et al., 2019), we did not include any outside points (i.e., control points) within 10 km of protected areas. Because protected areas are not distributed randomly, estimates of their effectiveness can be biased if there are systemic differences in baseline characteristics between protected and unprotected areas that affect both protection and disturbance (Butsic, Lewis, et al., 2017; Joppa & Pfaff, 2009). Commonly known as the "high and far" bias, protected areas are often concentrated in areas with high elevations, difficult accessibility, or low agricultural productivity (Joppa & Pfaff, 2009). To account for this nonrandom placement of protected areas, we selected in our subset of random pixels those that were similar in their underlying characteristics (e.g., remoteness, ruggedness, agricultural suitability) but differed in their protection status (i.e., treated vs. control). We used nearest-neighbor matching without replacement and a caliper size of a quarter of the estimated propensity score (Guo & Fraser, 2010).

We matched comparable observations of both groups (protected and unprotected) according to their propensity score. We derived this score from a logistic regression model in which treatment status was regressed on the length of the growing period, soil workability index, ruggedness, distance to the nearest major road, and distance to the nearest settlement. We assumed that the length of growing period (i.e., period of the year when the average temperature is ≥5°C and precipitation plus moisture storage exceeds half the potential evapotranspiration [IIASA & FAO, 2012]) and the soil workability (i.e., soil texture, effective soil depth and volume, and soil phases constraining soil management [IIASA & FAO, 2012]) determine the likelihood of rangeland productivity for use (Neal, 2024). Moreover, we assumed that ruggedness (i.e., topographically uneven, rocky or steep terrain [Sappington et al., 2007]) and distances from roads and settlements determine rangeland accessibility for use (Butsic, Munteanu, et al., 2017). Although elevation has been used as a matching variable in protected area impact evaluations (Joppa & Pfaff, 2009; Nelson & Chomitz, 2011), we did not include it due to its correlation with accessibility, terrain, and climate, which we considered separately and deemed more direct determinants of degradation than elevation. We resampled these control variables to our target resolution of 30 m with nearest neighbor resampling (Table 1). In the resulting matched data set of 68,030 rangeland pixels (34,394 inside, 33,636 outside protected areas), we assessed the standardized mean differences and variance ratios between protected and unprotected areas, which showed a balance in the matched sample (Appendices S2 & S3). Therefore, we were able to isolate the effect of protected area presence on rangeland green vegetation loss. We conducted the matching analysis with MatchIt (Ho et al., 2011) in R 1.1.453.

A second source of potential bias was unobservable static variables, such as weather fluctuations (Cameron & Trivedi, 2005), potentially affecting green vegetation cover in rangelands (Kohli et al., 2021). To account for the impact of drought, we used the annual standardized precipitation evapotranspiration index (SPEI-3) from the global SPEI database (https://spei.csic.es/database.html). The SPEI-3 is a cumulative drought index based on monthly precipitation and evapotranspiration data at 0.5° (ca. 50 km) resolution with an accumulation period of 3 months (SPEI-3) for 1988–2019, which represents aboveground productivity in grassland vegetation (Vicente-Serrano & National Center for Atmospheric Research Staff, 2010). Positive SPEI-3 values indicate wetter conditions and negative values indicate drier conditions for vegetation growth, with both extreme or prolonged deviations from zero indicating suboptimal conditions for vegetation growth. Although livestock grazing may also affect green vegetation cover, we did not include this variable (see below) in the model because of potential endogeneity problems from reverse causality (i.e., livestock cause green vegetation loss but corrals are placed in areas of high green vegetation cover).

We then parametrized a fixed-effects panel regression model and predicted the yearly logit-transformed rangeland green vegetation loss by treatment from 1988 to 2019. We chose the linear probability model over panel logit or probit specifications due to the difficulty of parameterizing the latter with fixed effects (Wooldridge, 2011). Specifically, we determined effectiveness with the following linear panel regression:

$$\text{logit}\,(Y_{it}) = \beta_1 \times P_i \text{year}_{it} + \beta_2 \times \text{SPEI}_{it} + e_{it}, \qquad (1)$$

where Y_{it} is rangeland greenness for i unit of observation and t point in time; P_i indicates whether an observation is protected (1, yes; 0, no); year_{it} is the time dummy variable indicating the

TABLE 1 Variables and data sets used to assess the effectiveness of protected areas on rangeland green vegetation loss in the southern Caucasus from 1988 to 2019.

Category	Variable	Unit	Period	Resolution	Source
Response	Green vegetation fraction	0–1 green vegetation cumulative endmember fraction	1988–2019	30 m	Lewińska et al., 2021
Treatment	Protected areas	1: protection 0: no protection	2020	Vector	World Database on Protected Areas (www.protectedplanet.com)
Panel regression variables	Administrative boundaries of countries	Armenia, Azerbaijan, Georgia, Iran, Turkey	Time invariant	Vector	Database of Global Administrative Areas (version 3.6) (www.gadm.org)
	International Union for Conservation of Nature management categories	Strictly protected (I & II), multiple-use areas (IV & V)	Time invariant	–	World Database on Protected Areas
	Size of protected areas	Area (km^2)	Time invariant	–	World Database on Protected Areas and WWF Caucasus Programme Office
	Age of protected areas	Years since establishment	1988–2020	–	World Database on Protected Areas and WWF Caucasus Programme Office
	Climatic influence	Standardized precipitation evapotranspiration index with an accumulation period of 3 months (SPEI-3)	1988–2018	0.5°	Vicente-Serrano et al., 2010
Control variables	Length of growing period	0–365 days	Time invariant	30 arcmin	IIASA & FAO, 2012
	Soil workability	Classes 0–7	Time invariant	30 arcmin	IIASA & FAO, 2012
	Ruggedness	0 (flat) to 1 (most rugged)	Time invariant	30 m	Calculated using a 90-m neighborhood rule (Sappington et al., 2007) based on topography data from Shuttle Radar Topography Mission (http://search.earthdata.nasa.gov)
	Distance to nearest major road	m	Time invariant	30 m	Open Street Map (www.openstreetmap.org)
	Distance to nearest settlement	m	Time invariant	30 m	Open Street Map and WWF Caucasus Programme Office

30-year study period; SPEI-3_{it} is climatic influences; rangeland greenness β_1 and β_2 are the coefficients to be estimated; and e_{it} is the error term.

To assess the impacts of contextual and management elements (Ghoddousi et al., 2022; Rodrigues & Cazalis, 2020), we classified the data set based on the country, management category, size, and age of protected areas and reran the fixed-effects panel regressions, comparing their protection effect in each run with the range-wide average. Each country has its own environmental policies and socioeconomic conditions that can affect protected area effectiveness (Butsic, Munteanu, et al., 2017). The management elements, such as the IUCN categories (strictly protected vs. multiple use), also reflect the levels of law enforcement and baseline land uses, important for protected area performance (Elleason et al., 2021). Furthermore, there is some evidence indicating positive impacts of protected area size and age on their effectiveness (Bowker et al., 2017; Wolf et al., 2021; Zhao et al., 2019). Accordingly, we stratified the analysis regarding interaction effects of treatment and country (C_i), IUCN categories (IUCN_i), size (s_{it}), and age of protected area (a_{it}):

$$\mathrm{logit}\,(Y_{it}) = \beta_1 \times P_i\mathrm{year}_{it} + \beta_2 \times \mathrm{SPEI}_{it} + \beta_3 \times C_i + \beta_4 \times \mathrm{IUCN}_i + \beta_5 \times s_{it} + \beta_6 \times a_{it} + e_{it}. \quad (2)$$

We performed panel regressions with the plm package (Croissant & Millo, 2008).

Livestock grazing pressure

We assessed the relationship between rangeland green vegetation loss and livestock (i.e., sheep, goat, and cattle) presence in protected areas to quantify whether livestock grazing pressure reduced the green vegetation (Jamsranjav et al., 2018). Transhumance by using summer and winter pastures is common in the Caucasus (de Leeuw et al., 2019; Neudert, 2015; Wiesmair et al., 2016), and the location of livestock corrals is a proxy for livestock presence (Bleyhl et al., 2019). Wild large herbivores (e.g., bezoar goat, mouflon) occur in small numbers inside some protected areas of the Caucasus but are typically absent on unprotected lands (Bleyhl et al., 2019; Kuemmerle et al., 2020) and, thus, are not a main cause of overgrazing and degradation. We stratified the region to elevations above 1500 m asl with a topography model (EROS Center, 2018) to identify summer pastures (Bleyhl et al., 2019; de Leeuw et al., 2019). We used the most recent high-resolution satellite images available in 2020 in Google Earth (https://earth.google.com) to digitize livestock corrals for all summer pastures of the southern Caucasus. We identified the corrals as artificial structures (e.g., shack, stone wall, or tarp) in open pastures with homogenous open soil near the corral clearly visible in high-resolution imagery (Bleyhl et al., 2019). We did not analyze winter pastures because they are commonly located in more accessible, lower elevation areas coinciding with a variety of other human infrastructures that are easily misidentified as corrals. To facilitate the digitizing, we used a systematic grid of 1.5-km resolution over the study area. We then extracted the number of corrals in our protected areas (see above) and calculated their density in summer pastures of protected areas. Four protected areas, all in Azerbaijan (Shamkir, Korchay [I & IV], and Hirkan [IV]), were at <1500 m asl, and we removed them from this part of our analyses. As a measure of degradation, we calculated the mean green vegetation CEF loss from 2014 to 2019 in each protected area. We used this period because it corresponded to when active corrals were observable in Google Earth satellite images. We were unable to perform a trend analysis on the corral numbers because data from older periods were not available. However, because the location of corrals is stable in the region, we are confident that their contemporary location is a robust proxy for livestock grazing pressure. We used Pearson's correlation coefficient test to assess the relationship between mean green vegetation loss and corral density. We further tested this relationship for different subsets of our data based on the country and management categories (strictly protected vs. multiple-use areas).

RESULTS

State of rangelands in the southern Caucasus

Protected areas in the southern Caucasus were predominantly covered by rangelands, with >50% of protected areas covered by rangelands in 36 out of the 52 protected areas. Only one protected area in Georgia (Mtirala National Park) did not contain any rangeland, and we removed it from the analyses. There was no clear overarching trend in the green vegetation CEF from 1988 to 2019. Instead, green vegetation CEF across rangelands in the southern Caucasus was highly dynamic inside and outside protected areas (0–0.87).

Rangelands in Georgia had the highest mean green vegetation CEF level (0.25) and variation (0.01), whereas mean green vegetation CEF level and variation were lower in Azerbaijan (0.13 and 0.008, respectively) and Iran (0.11 and 0.008, respectively). Across the study area, the highest absolute number of pixels with a green vegetation loss signal was in Turkey. We did not find evidence that the observed changes in rangeland vegetation were due to changes in weather conditions because the temporal dynamics of rangeland green vegetation CEF did not follow SPEI-3 patterns (Appendices S4–S8), and our linear regression models showed limited explanatory power of SPEI-3 to explain variability in green vegetation CEF.

TABLE 2 The comparison of the country-level and International Union for Conservation of Nature (IUCN) management category deviations from the average effect (−0.013) of protection on rangeland green vegetation loss in the southern Caucasus.

Category	Protection effect	SE
Country		
Armenia	−0.030	<0.001
Azerbaijan	−0.018	<0.001
Georgia	0.014	<0.001
Iran	−0.007	<0.001
Turkey	0.010	<0.001
IUCN		
I & II	−0.008	<0.001
IV & V	−0.015	<0.001

Protected area effectiveness

Across the southern Caucasus, protection decreased rangeland green vegetation CEF by 0.013 (standard error of difference <0.001) compared with pixels outside them, suggesting that protected areas were ineffective in preventing green vegetation loss. In Armenia, Azerbaijan, and Iran, matched pixels outside protected areas had higher green vegetation CEF than those inside protected areas (Figure 3). This trend was reversed in Georgia and Turkey (Figure 3), and protected rangelands in the northern parts of the study area had the highest mean green vegetation CEF (e.g., in Borjomi National Park, Georgia and Posof Sanctuary, Turkey) (Figure 4).

Impact of contextual and management elements

The impact of protected areas in preventing rangeland green vegetation loss differed over space and time, as well as among countries and management categories (Table 2). Protected areas in Armenia were the most ineffective, followed by Azerbaijan. In Iran, their impact was similar to the range-wide average (−0.013), and in Georgia and Turkey, protected areas were effective according to our assessment.

Comparing the protected area impact across management categories, we found lower effectiveness for multiple-use protected areas (IUCN categories IV & V). The effects of strictly protected areas (IUCN categories I & II) were similar to the range-wide average (−0.013) (Table 2). We found no effect of protected area size and age on rangeland greenness.

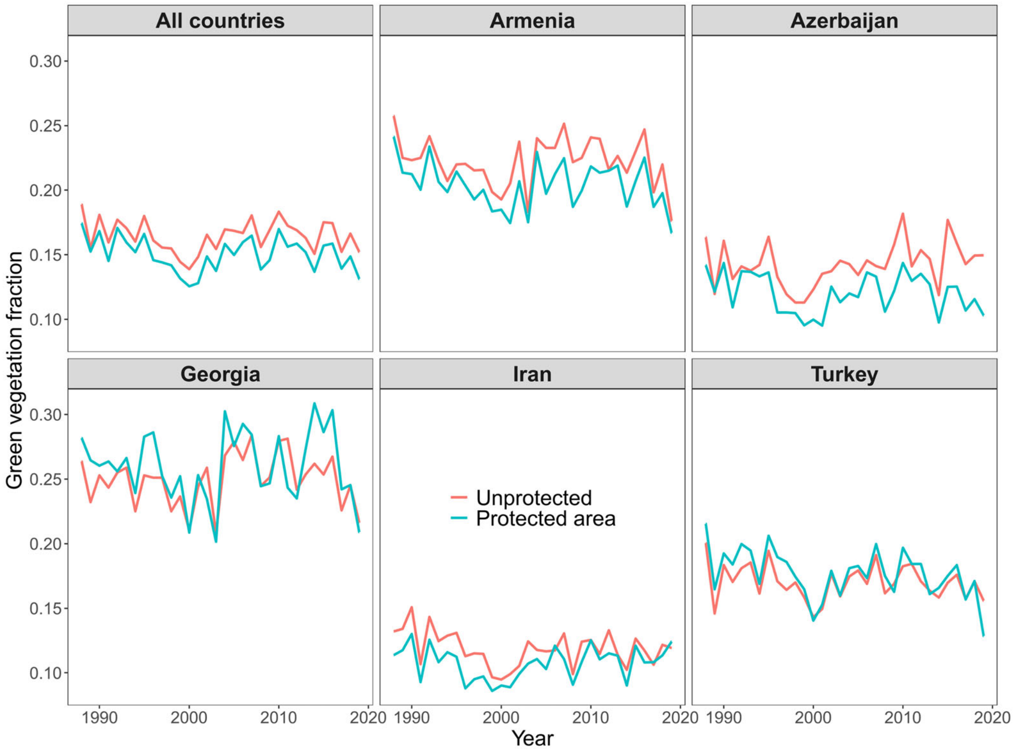

FIGURE 3 Trajectories of country-level mean rangeland green vegetation in matched protected and unprotected pixels over time (1988–2019) in the southern Caucasus. Green vegetation was measured using the cumulative endmember fraction derived through spectral unmixing.

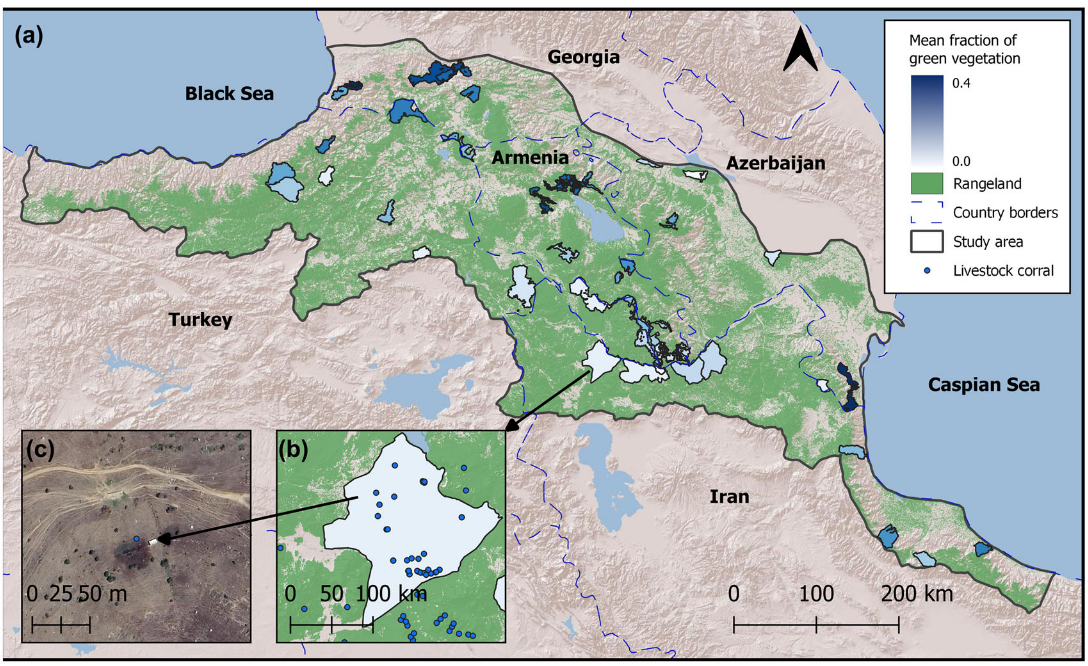

FIGURE 4 Livestock pressure and green vegetation loss in the southern Caucasus: (a) green vegetation trends in protected areas of the southern Caucasus from 1988 to 2019, (b) distribution of livestock corrals in summer pastures (>1500 m asl) inside and outside a protected area, and (c) an example of one livestock corral in Google Earth.

Impacts of livestock grazing on rangeland green vegetation loss

We digitized 10,116 corrals (Armenia 1374, Azerbaijan 1361, Georgia 798, Iran 3181, Turkey 3402). Livestock corrals were widespread with high concentrations in the Gegham Mountains in Armenia, in the northern foothills of Murovdag in Azerbaijan, in the Javakheti Plateau in Georgia, along the northern slopes of Alborz Mountains and the foothills of Mt. Sabalan in Iran, in the Yalnizçam Mountains, and in the foothills of Mt. Ararat or Agri and Mt. Kisir in Turkey. The highest densities in rangelands occurred in Iran (0.05 corral/km^2), Turkey (0.04), and Armenia (0.04) and the lowest in Azerbaijan (0.02). Among protected areas, we found the highest density of livestock corrals in Iranian protected areas (0.30). In other countries, corral density was relatively similar inside protected areas (Armenia and Turkey 0.04 corral/km^2, Georgia 0.05 corral/km^2, Azerbaijan 0.06 corral/km^2). There was a much higher density of livestock corrals in multiple-use protected areas (0.12 corral/km^2 in 31 protected areas) than in strictly protected areas (0.04 corral/km^2 in 17 protected areas).

We found a significant ($p < 0.05$ threshold) negative relationship between mean green vegetation loss from 2014 to 2019 and the density of corrals in protected areas ($r = -0.42$, $p < 0.01$). However, this relationship varied among countries. In Armenia ($r = -0.02$, $p = 0.92$) and Georgia ($r = -0.11$, $p = 0.80$), we observed weak and nonsignificant relationships between mean green vegetation loss and the density of livestock corrals. In protected areas in Iran, there was a nonsignificant negative correlation ($r = -0.46$, $p = 0.20$) between these 2 variables. We detected the strongest negative relationships in Turkey (although nonsignificant) ($r = -0.68$, $p = 0.06$) and Azerbaijan ($r = -0.85$, $p = 0.02$). When considering the management categories, there was a negative relationship between mean green vegetation loss from 2014 to 2019 and the density of corrals for both strictly protected ($r = -0.39$, $p = 0.10$) and multiple-use protected areas ($r = -0.40$, $p = 0.02$), although the former relationship was not significant.

DISCUSSION

Land degradation affects over one third of the global land mass (FAO & ITPS, 2015) and is expected to further increase due to climate change and intensifying land use (Stanimirova et al., 2019). Rangelands, covering a quarter of Earth's surface and directly supporting over one billion people, are particularly affected by degradation (Bond & Parr, 2010; Strömberg & Staver, 2022). However, the effectiveness of protected areas in abating rangeland degradation remains unclear despite many protected areas in rangeland regions. Here, we use satellite-based measures of green vegetation loss in rangelands (Lewińska et al., 2021) as a degradation proxy in a robust impact evaluation framework to assess protected area effectiveness in the southern Caucasus. Three main findings were derived from our assessment of 52 protected areas in five countries. First, protected areas in rangelands were largely ineffective in reducing green vegetation loss from 1988 to 2019, a pattern contrasting results found for assessments of the effectiveness of protected areas in safeguarding forests in the Caucasus and other post-Soviet countries (Bragina et al., 2015; Butsic, Munteanu, et al., 2017). Second, the level of effectiveness of protected areas varied with contextual and management elements such as country differences (protection only effective in Georgia and Turkey) or protected area management (multiple-use areas less effective than strictly protected areas). Third, through mapping livestock corrals as an indicator of grazing pressure, we could link the observed green vegetation loss signals to livestock grazing inside protected areas, especially in Azerbaijan and Turkey. This suggests overgrazing is a main driver of the observed ineffectiveness. Our approach is transferable and can provide insight into land-use pressures on protected rangelands more generally.

Our first primary finding was the apparent ineffectiveness of protected areas in reducing rangeland degradation in the southern Caucasus. Surprisingly, and despite some spatiotemporal variation across the region, rangelands outside protected areas had an overall higher green vegetation fraction compared with matched locations inside. This result was unexpected because protected areas are meant to reduce degradation and improve the state of ecosystems over time. However, there are at least 3 explanations for this finding.

First, this pattern could be due to agricultural intensification in farmland areas outside protected areas. After decades of agricultural expansion during the Soviet era, the post-Soviet transition and armed conflicts led to widespread abandonment of farmlands (Baumann et al., 2014; Buchner et al., 2020, 2022). Formerly fertilized but now abandoned land outside protected areas is likely to have a higher green vegetation level than more marginal and historically unfertilized areas inside protected areas. Unfortunately, due to a lack of data on the levels of fertilizers or the initial state of vegetation, we were unable to test these factors in our analyses.

Second, most protected areas in the Caucasus, even with the highest protection levels (i.e., IUCN categories I & II), allow livestock grazing inside their boundaries because it often predates park establishments. This allowance can create goodwill between local people and the parks. According to our data, the densities of livestock corrals in summer pastures of protected areas were similar to pastures outside (in Armenia and Turkey) or even higher than in unprotected areas (in Azerbaijan ×3, Georgia ×1.6, and Iran ×6). Pastures inside protected areas are thus de facto a common resource pool in many parts of the southern Caucasus, making them vulnerable to overexploitation when institutions weaken (Dietz et al., 2003; Hardin, 1968; Ostrom et al., 1999). This was, unfortunately, common after the breakdown of the Soviet Union and when armed conflicts prevailed.

Third, the security and infrastructure provided by protected areas may be desired by pastoralists (A.G. personal observation) and lead to higher livestock densities there. For example, in many protected areas, forestry or ranger roads allow easier access to high pastures for pastoralists, and patrolling rangers reduce the risk of livestock theft. Therefore, despite some restrictions, protected areas may have a limited impact on overgrazing and degradation.

Our second primary finding underlines that contextual and management elements greatly affect protected areas' effectiveness. Importantly, our results are consistent with previous studies showing notable variations in the performance of protected areas across countries (Elleason et al., 2021; Geldmann et al., 2019). Encouragingly, despite more pronounced green vegetation loss across the country, a high density of livestock corrals, and strong land-use pressure (Kurdoğlu & Çokçaliskan, 2011), protected areas of Turkey were overall effective in our assessment. Similarly, protected areas in Georgia were effective in protecting their rangelands against degradation and had a higher green vegetation level than unprotected areas. However, in other countries, protected areas were ineffective against rangeland green vegetation loss. Despite an increasing green vegetation trend (Lewińska et al., 2021), protected areas in Armenia had limited effectiveness in avoiding green vegetation loss. In Azerbaijan, the overall degradation of rangelands was observed previously (Lewińska et al., 2021; Neudert et al., 2013), which, together with an increase in livestock numbers (de Leeuw et al., 2019), may explain the lower effectiveness we found. Finally, the Iranian protected areas in our study area were also ineffective. These protected areas currently lack sufficient resources to curb human pressures, partly due to economic sanctions (Ghoddousi et al., 2017; Khalatbari et al., 2018). The variations we observed suggest a need for conservation initiatives to move beyond mere protected area expansion targets and seek to improve the effectiveness of existing protected areas as well.

A key determinant of the effectiveness of protected areas is their governance (Eklund & Cabeza, 2017). However, the comparison of the performance of protected areas under different management categories has been inconclusive. For example, multiple-use areas were more effective than strictly protected areas in preventing tropical forest loss due to fire (Nelson & Chomitz, 2011). Conversely, strictly protected areas were more effective against agricultural expansion into their boundaries (Joppa & Pfaff, 2011). Finally, across the globe, differences in governance types had limited power in explaining the human footprint inside protected areas (Elleason et al., 2021). In our study, the effectiveness of strictly protected areas (IUCN categories I & II) was higher than that of multiple-use areas (IUCN categories IV–VI). This pattern was somewhat expected because a more diverse set of land uses (e.g., livestock pastoralism, agriculture, mining) is permitted in multiple-use protected areas but might also hint to some of these land uses leading to degradation. Despite the existence of livestock quota, grazing season, and other management measures in most of the multiple-use protected areas, these regulations are often not fully enforced in the region (A.G. personal observation). Setting socially acceptable and ecologically sustainable levels of land use would be needed to avoid further damage to these landscapes. More generally, our results showed that the strength of institutions, whether at country or site level, plays a major role in the effectiveness of protected areas in the southern Caucasus.

Our third primary finding was that livestock grazing appears to be the likely cause for the low effectiveness of rangeland protected areas. Generally, livestock, occupying around 26% of the world's terrestrial area (Foley et al., 2011; Godde et al., 2018), is a major driver of habitat degradation (Bond & Parr, 2010). Our analyses showed a clear relationship between livestock presence and rangeland degradation, approximated by green vegetation loss inside protected areas, in line with other studies, especially in Azerbaijan (de Leeuw et al., 2019; Lewińska et al., 2021). We found strong country-specific variation in this association, with the strongest relationships in Azerbaijan and Turkey. This might reflect environmental variation, for example, drier conditions that are more prone to overgrazing (Mysterud, 2006; Neudert et al., 2013) and differences in stocking rates and livestock husbandry regimes (e.g., Azerbaijan had a stock of 7.57 million sheep vs. 0.61 million in Armenia in 2019; FAO, 2023). Moreover, the collapse of the Soviet Union resulted in marked changes in livestock numbers and husbandry systems in the former Soviet Caucasus countries (all except for Iran and Turkey). Plummeting livestock numbers in all countries but Azerbaijan (FAO, 2023), more stationary husbandry regimes (Wiesmair et al., 2016), and disrupted transhumance (Radvanyi & Muduyev, 2007) have redistributed grazing pressure and resulted in degradation in some areas and recovery in others (Dara et al., 2020; de Leeuw et al., 2019; Neudert, 2015). Given such variation in the impacts of livestock pastoralism on rangelands, a social–ecological understanding of this land use in protected areas is crucial (de Leeuw et al., 2019; Jamsranjav et al., 2018).

Our results highlight that despite a higher density of livestock corrals in multiple-use areas, the relationship between degradation levels and livestock presence is similar across different protected area management categories. This pattern, in combination with the higher effectiveness of strictly protected areas, shows that livestock grazing, wherever at higher levels, is a major driver of rangeland degradation in this region. Unsustainable levels of livestock grazing cause soil degradation, plant biomass reduction, and desertification (Hilker et al., 2014). In Azerbaijan, Neudert et al. (2013) documented overstocking only in some pastures; in the majority of others, spatially unadjusted grazing caused degradation, which indicates the need for destocking and changing herding patterns. However, determining sustainable levels of grazing is challenging because overgrazing is often considered a value-laden determination, reflecting management perspectives that may conflict with those of local people (Mysterud, 2006). Therefore, benefitting from local traditional knowledge and robust assessments of green vegetation dynamics is necessary to devise site-specific grazing management regimes that reconcile conservation and development goals, especially in multiple-use protected areas.

Our results strengthen calls for moving beyond merely using forest cover as an indicator of protected area effectiveness because it may lead to an oversimplified or even misleading picture (Ghoddousi et al., 2022; Green et al., 2020; Htun et al., 2009). This is particularly crucial for regions such as the Caucasus, where open landscapes are widespread, and relying on forest cover as the only indicator of performance may lead to the assumption that protected areas are effective because forest loss is limited overall (Bragina et al., 2015; Buchner et al., 2020). Furthermore, in this region, land use often does not result in complete conversion (Buchner et al., 2020) but more commonly leads to habitat degradation (Cortner et al., 2024). For example, around 20% of rangelands in the Caucasus experienced degradation by green vegetation loss from 1987 to 2019 (Lewińska et al., 2021), whereas conversion of rangelands was limited (Buchner et al., 2020). During the same period, the area of forest degradation in Georgia was substantially larger than the area of deforestation (ca. >3500 km^2 vs. about 160 km^2, respectively) (Chen et al., 2021). Given these tendencies, further development of habitat degradation indices and integration of them in protected area impact evaluation frameworks are essential.

Monitoring rangeland degradation is not straightforward due to methodological and definitional constraints (Dubovyk, 2017; Gibbs & Salmon, 2015; Vogt et al., 2011). We used CEFs from satellite image time series, an approach well suited for landscapes with sparse vegetation (Lewińska et al., 2020). We additionally identified livestock corrals as an indicator of grazing pressure, highlighting the usefulness of remote sensing in assessing the effectiveness of nonforested protected areas. Despite these advances, we acknowledge some limitations. First, there may be other drivers of rangeland green vegetation loss (e.g., fire, mowing) we did not consider. Second, these drivers, including livestock grazing, may be compatible with protected area goals in some cases, depending on the intensity (Gavin et al., 2018). However, livestock overgrazing is a major threat to rangelands in the Caucasus (de Leeuw et al., 2019; Neudert, 2021) and is of concern in its protected areas. Although the density of livestock corrals was related to the overall level of green vegetation loss in protected areas, this data set does not consider livestock type and numbers, both crucial determinants of grazing pressure. Moreover, some of the corrals may be temporally inactive or abandoned, which we could not detect from satellite images. The exclusion of winter corrals from the analysis due to difficulties in detecting them may have biased our assessment of livestock grazing pressure if transhumance is not exercised. However, pastoralism in the Caucasus predominantly involves migrating between summer and winter pastures (de Leeuw et al., 2019); therefore, summer corrals are a suitable proxy for livestock grazing pressure in this region. Finally, the identification of summer pastures as rangelands >1500 m asl might omit some summer pastures, although another study (de Leeuw et al., 2019) also indicated that summer pastures in the Caucasus are above 1600 m asl.

Land use is a major threat to biodiversity, and protected areas are key to limiting this threat (Schulze et al., 2018). Our study is among the first comprehensive assessments of protected area effectiveness in protecting rangelands (Mammides et al., 2024). The methodology we devise can be applied to many predominantly open landscapes worldwide. Our approach allows for a better consideration of rangeland grazing pressure and socioeconomic and contextual elements to ensure the social and ecological outcomes of protected areas (Ghoddousi et al., 2022; Pozo et al., 2021). Moreover, the overall tendency of rangeland recovery in this region (Lewińska et al., 2021) and the move toward a more sedentary livestock husbandry, instead of nomadic pastoralism (Wiesmair et al., 2016), suggest there may be opportunities for restoration of the ecological function of unprotected rangelands and for the conservation and reintroduction of wild herbivores (e.g., mouflon, goitered gazelle [*Gazella subgutturosa*]), when combined with poaching alleviation (Bleyhl et al., 2019; Kuemmerle et al., 2020).

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REFERENCES

Bai, Z. G., Dent, D. L., Olsson, L., & Schaepman, M. E. (2008). Proxy global assessment of land degradation. *Soil Use and Management*, *24*, 223–234.

Banks-Leite, C., Ewers, R. M., Folkard-Tapp, H., & Fraser, A. (2020). Countering the effects of habitat loss, fragmentation, and degradation through habitat restoration. *One Earth*, *3*, 672–676.

Baumann, M., Radeloff, V. C., Avedian, V., & Kuemmerle, T. (2014). Land-use change in the Caucasus during and after the Nagorno-Karabakh conflict. *Regional Environmental Change*, *15*, 1703–1716.

Bleyhl, B., Arakelyan, M., Askerov, E., Bluhm, H., Gavashelishvili, A., Ghasabian, M., Ghoddousi, A., Heidelberg, A., Khorozyan, I., Malkhasyan, A., Manvelyan, K., Masoud, M., Moqanaki, E. M., Radeloff, V. C., Soofi, M., Weinberg, P., Zazanashvili, N., & Kuemmerle, T. (2019). Assessing niche overlap between domestic and threatened wild sheep to identify conservation priority areas. *Diversity and Distributions*, *25*, 129–141.

Bleyhl, B., Baumann, M., Griffiths, P., Heidelberg, A., Manvelyan, K., Radeloff, V. C., Zazanashvili, N., & Kuemmerle, T. (2017). Assessing landscape connectivity for large mammals in the Caucasus using Landsat 8 seasonal image composites. *Remote Sensing of Environment*, *193*, 193–203.

Bond, W. J., & Parr, C. L. (2010). Beyond the forest edge: Ecology, diversity and conservation of the grassy biomes. *Biological Conservation*, *143*, 2395–2404.

Bowker, J. N., De Vos, A., Ament, J. M., & Cumming, G. S. (2017). Effectiveness of Africa's tropical protected areas for maintaining forest cover. *Conservation Biology*, *31*, 559–569.

Bragina, E. V., Radeloff, V. C., Baumann, M., Wendland, K., Kuemmerle, T., & Pidgeon, A. M. (2015). Effectiveness of protected areas in the Western Caucasus before and after the transition to post-socialism. *Biological Conservation*, *184*, 456–464.

Brandt, R. (1992). Soviet environment slips down the agenda. *Science*, *255*, 22–23.

Bruner, A. G., Gullison, R. E., Rice, R. E., & Da Fonseca, G. A. (2001). Effectiveness of parks in protecting tropical biodiversity. *Science*, *291*, 125–128.

Buchner, J., Butsic, V., Yin, H., Kuemmerle, T., Baumann, M., Zazanashvili, N., Stapp, J., & Radeloff, V. C. (2022). Localized versus wide-ranging effects of the post-Soviet wars in the Caucasus on agricultural abandonment. *Global Environmental Change*, *76*, Article 102580.

Buchner, J., Yin, H., Frantz, D., Kuemmerle, T., Askerov, E., Bakuradze, T., Bleyhl, B., Elizbarashvili, N., Komarova, A., Lewińska, K. E., Rizayeva, A., Sayadyan, H., Tan, B., Tepanosyan, G., Zazanashvili, N., & Radeloff, V. C. (2020). Land-cover change in the Caucasus Mountains since 1987 based on the topographic correction of multi-temporal Landsat composites. *Remote Sensing of Environment*, *248*, Article 111967.

Butsic, V., Lewis, D. J., Radeloff, V. C., Baumann, M., & Kuemmerle, T. (2017). Quasi-experimental methods enable stronger inferences from observational data in ecology. *Basic and Applied Ecology*, *19*, 1–10.

Butsic, V., Munteanu, C., Griffiths, P., Knorn, J., Radeloff, V. C., Lieskovský, J., Mueller, D., & Kuemmerle, T. (2017). The effect of protected areas on forest disturbance in the Carpathian Mountains 1985–2010. *Conservation Biology*, *31*, 570–580.

Cameron, A. C., & Trivedi, P. K. (2005). *Microeconometrics: Methods and applications*. Cambridge University Press.

Chen, S., Woodcock, C. E., Bullock, E. L., Arévalo, P., Torchinava, P., Peng, S., & Olofsson, P. (2021). Monitoring temperate forest degradation on Google Earth Engine using Landsat time series analysis. *Remote Sensing of Environment*, *265*, Article 112648.

Convention on Biological Diversity (CBD). (2022). *Decision adopted by the Conference of the Parties to the Convention on Biological Diversity: 15.4. CBD/COP/DEC/15/4*. Conference of the Parties to the Convention on Biological Diversity.

Cortner, O., Chen, S., Olofsson, P., Gollnow, F., Torchinava, P., & Garrett, R. D. (2024). Exploring natural and social drivers of forest degradation in post-Soviet Georgia. *Global Environmental Change*, *84*, Article 102775.

Croissant, Y., & Millo, G. (2008). Panel data econometrics in R: The plm package. *Journal of Statistical Software*, *27*, 1–43.

Dara, A., Baumann, M., Freitag, M., Hölzel, N., Hostert, P., Kamp, J., Müller, D., Prishchepov, A. V., & Kuemmerle, T. (2020). Annual Landsat time series reveal post-Soviet changes in grazing pressure. *Remote Sensing of Environment*, *239*, Article 111667.

de Jong, R., de Bruin, S., de Wit, A., Schaepman, M. E., & Dent, D. L. (2011). Analysis of monotonic greening and browning trends from global NDVI time-series. *Remote Sensing of Environment*, *115*, 692–702.

de Leeuw, J., Rizayeva, A., Namazov, E., Bayramov, E., Marshall, M. T., Etzold, J., & Neudert, R. (2019). Application of the MODIS MOD 17 Net Primary Production product in grassland carrying capacity assessment. *International Journal of Applied Earth Observation and Geoinformation*, *78*, 66–76.

Dietz, T., Ostrom, E., & Stern, P. C. (2003). The struggle to govern the commons. *Science*, *302*, 1907–1912.

Dubovyk, O. (2017). The role of Remote Sensing in land degradation assessments: Opportunities and challenges. *European Journal of Remote Sensing*, *50*, 601–613.

Dudley, N., Eufemia, L., Fleckenstein, M., Periago, M. E., Petersen, I., & Timmers, J. F. (2020). Grasslands and savannahs in the UN Decade on Ecosystem Restoration. *Restoration Ecology*, *28*, 1313–1317.

Earth Resources Observation and Science (EROS) Center. (2018). *USGS EROS Archive - Digital Elevation - Shuttle Radar Topography Mission (SRTM) 1 Arc-Second Global*. Author.

Eklund, J., & Cabeza, M. (2017). Quality of governance and effectiveness of protected areas: Crucial concepts for conservation planning. *Annals of the New York Academy of Sciences*, *1399*, 27–41.

Elleason, M., Guan, Z., Deng, Y., Jiang, A., Goodale, E., & Mammides, C. (2021). Strictly protected areas are not necessarily more effective than areas in which multiple human uses are permitted. *Ambio*, *50*, 1058–1073.

Elmore, A. J., Mustard, J. F., Manning, S. J., & Lobell, D. B. (2000). Quantifying vegetation change in semiarid environments: Precision and accuracy of spectral mixture analysis and the normalized difference vegetation index. *Remote Sensing of Environment*, *73*, 87–102.

European Commission JRC. (2018). *Digital Observatory for Protected Areas*. http://data.europa.eu/89h/jrc-dopa-maps-and-datasets

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., … Zaks, D. P. M. (2011). Solutions for a cultivated planet. *Nature*, *478*, 337–342.

Food and Agriculture Organization of the United Nations (FAO), & Intergovernmental Technical Panel on Soils (ITPS). (2015). *Status of the world's soil resources (SWSR)–main report*. Author.

Food and Agriculture Organization of the United Nations (FAO). (2023). *FAOSTAT*. Author.

Frantz, D., Hostert, P., Rufin, P., Ernst, S., Röder, A., & van der Linden, S. (2022). Revisiting the past: Replicability of a historic long-term vegetation dynamics assessment in the era of big data analytics. *Remote Sensing*, *14*, Article 597.

Fuller, C., Ondei, S., Brook, B. W., & Buettel, J. C. (2019). First, do no harm: A systematic review of deforestation spillovers from protected areas. *Global Ecology and Conservation*, *18*, Article e00591.

Gavin, M. C., McCarter, J., Berkes, F., Sterling, E. J., & Turner, N. J. (2018). Protected land: Many factors shape success. *Science*, *361*, 561–561.

Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M., & Burgess, N. D. (2013). Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation*, *161*, 230–238.

Geldmann, J., Manica, A., Burgess, N. D., Coad, L., & Balmford, A. (2019). A global-level assessment of the effectiveness of protected areas at resisting anthropogenic pressures. *Proceedings of the National Academy of Sciences of the United States of America*, *116*(46), 23209–23215.

Ghoddousi, A., Egli, L., Soofi, M., Khorozyan, I., & Waltert, M. (2017). After sanctions: The urge to upgrade and integrate conservation in Iran. *Frontiers in Ecology and the Environment*, *15*, 9–10.

Ghoddousi, A., Loos, J., & Kuemmerle, T. (2022). An outcome-oriented, social–ecological framework for assessing protected area effectiveness. *Bioscience*, *72*, 201–212.

Gibbs, H. K., & Salmon, J. M. (2015). Mapping the world's degraded lands. *Applied Geography*, *57*, 12–21.

Godde, C. M., Garnett, T., Thornton, P. K., Ash, A. J., & Herrero, M. (2018). Grazing systems expansion and intensification: Drivers, dynamics, and trade-offs. *Global Food Security*, *16*, 93–105.

Green, E. J., McRae, L., Freeman, R., Harfoot, M. B. J., Hill, S. L. L., Baldwin-Cantello, W., & Simonson, W. D. (2020). Below the canopy: Global trends in forest vertebrate populations and their drivers. *Proceedings of the Royal Society B: Biological Sciences*, *287*, Article 20200533.

Guan, Z., Elleason, M., Goodale, E., & Mammides, C. (2021). Global patterns and potential drivers of human settlements within protected areas. *Environmental Research Letters*, *16*, Article 064085.

Gunya, A., Lysenko, A., Lysenko, I., & Mitrofanenko, L. (2021). Transformation of nature protection institutions in the North Caucasus: From a state monopoly of governance to multi-actor management. *Sustainability*, *13*, Article 12145.

Guo, S., & Fraser, M. W. (2010). *Propensity score analysis: Statistical methods and applications*. SAGE Publications.

Hardin, G. (1968). The tragedy of the commons. *Science*, *162*, 1243–1248.

Hilker, T., Natsagdorj, E., Waring, R. H., Lyapustin, A., & Wang, Y. (2014). Satellite observed widespread decline in Mongolian grasslands largely due to overgrazing. *Global Change Biology*, *20*, 418–428.

Ho, D., Imai, K., King, G., & Stuart, E. A. (2011). MatchIt: Nonparametric preprocessing for parametric causal inference. *Journal of Statistical Software*, *42*, 1–28.

Hostert, P., Röder, A., & Hill, J. (2003). Coupling spectral unmixing and trend analysis for monitoring of long-term vegetation dynamics in Mediterranean rangelands. *Remote Sensing of Environment*, *87*, 183–197.

Htun, N. Z., Mizoue, N., Kajisa, T., & Yoshida, S. (2009). Deforestation and forest degradation as measures of Popa Mountain Park (Myanmar) effectiveness. *Environmental Conservation*, *36*, 218–224.

Huete, A. R., Jackson, R. D., & Post, D. F. (1985). Spectral response of a plant canopy with different soil backgrounds. *Remote Sensing of Environment*, *17*, 37–53.

International Institute for Applied Systems Analysis (IIASA), & Food and Agriculture Organization of the United Nations (FAO). (2012). *Global Agro-ecological Zones (GAEZ v3.0)*. Author.

International Livestock Research Institute (ILRI), International Union for Conservation of Nature (IUCN), Food and Agriculture Organization (FAO), World Wildlife Fund (WWF), UN Environment Programme (UNEP), & International Land Coalition (ILC). (2021). *Rangelands Atlas*. ILRI.

Jamsranjav, C., Reid, R. S., Fernández-Giménez, M. E., Tsevlee, A., Yadamsuren, B., & Heiner, M. (2018). Applying a dryland degradation framework for rangelands: The case of Mongolia. *Ecological Applications*, *28*, 622–642.

Jones, K. R., Venter, O., Fuller, R. A., Allan, J. R., Maxwell, S. L., Negret, P. J., & Watson, J. E. M. (2018). One-third of global protected land is under intense human pressure. *Science*, *360*, 788–791.

Joppa, L. N., & Pfaff, A. (2009). High and far: Biases in the location of protected areas. *PLoS ONE*, *4*, Article e8273.

Joppa, L. N., & Pfaff, A. (2011). Global protected area impacts. *Proceedings of the Royal Society B: Biological Sciences*, *278*, 1633–1638.

Khalatbari, L., Brito, J. C., Ghoddousi, A., Abolghasemi, H., Breitenmoser, U., Breitenmoser-Würsten, C., Yusefi, G. H., Ostrowski, S., & Durant, S. M. (2018). Sanctioning to extinction in Iran. *Science*, *362*, Article 1255.

Kohli, M., Mijiddorj, T. N., Suryawanshi, K. R., Mishra, C., Boldgiv, B., & Sankaran, M. (2021). Grazing and climate change have site-dependent interactive effects on vegetation in Asian montane rangelands. *Journal of Applied Ecology*, *58*, 539–549.

Krever, V., Zazanashvili, N., Jungius, H., Williams, L., & Petelin, D. (2001). *Biodiversity of the Caucasus ecoregion: An analysis of biodiversity and current threats and initial investment portfolio*. World Wide Fund for Nature.

Kuemmerle, T., Bluhm, H., Ghoddousi, A., Arakelyan, M., Askerov, E., Bleyhl, B., Ghasabian, M., Gavashelishvili, A., Heidelberg, A., Malkhasyan, A., Manvelyan, K., Soofi, M., Yarovenko, Y., Weinberg, P., & Zazanashvili, N. (2020). Identify priority areas for restoring mountain ungulates in the Caucasus. *Conservation Science and Practice*, *2*, Article e276.

Kuemmerle, T., Röder, A., & Hill, J. (2006). Separating grassland and shrub vegetation by multidate pixel-adaptive spectral mixture analysis. *International Journal of Remote Sensing*, *27*, 3251–3271. https://www.tandfonline.com/doi/full/10.1080/01431160500488944

Kurdoğlu, O., & Çokçaliskan, B. A. (2011). Assessing the effectiveness of protected area management in the Turkish Caucasus. *African Journal of Biotechnology*, *10*, 17208–17222.

Laurance, W. F., Carolina Useche, D., Rendeiro, J., Kalka, M., Bradshaw, C. J. A., Sloan, S. P., Laurance, S. G., Campbell, M., Abernethy, K., Alvarez, P., Arroyo-Rodriguez, V., Ashton, P., Benítez-Malvido, J., Blom, A., Bobo, K. S., Cannon, C. H., Cao, M., Carroll, R., Chapman, C., … Zamzani, F. (2012). Averting biodiversity collapse in tropical forest protected areas. *Nature*, *489*, 290–294.

Leberger, R., Rosa, I. M. D., Guerra, C. A., Wolf, F., & Pereira, H. M. (2020). Global patterns of forest loss across IUCN categories of protected areas. *Biological Conservation*, *241*, Article 108299.

Leroux, S. J., Krawchuk, M. A., Schmiegelow, F., Cumming, S. G., Lisgo, K., Anderson, L. G., & Petkova, M. (2010). Global protected areas and IUCN designations: Do the categories match the conditions? *Biological Conservation*, *143*, 609–616.

Lewińska, K. E., Buchner, J., Bleyhl, B., Hostert, P., Yin, H., Kuemmerle, T., & Radeloff, V. C. (2021). Changes in the grasslands of the Caucasus based on Cumulative Endmember Fractions from the full 1987–2019 Landsat record. *Science of Remote Sensing*, *4*, Article 100035.

Lewińska, K. E., Hostert, P., Buchner, J., Bleyhl, B., & Radeloff, V. C. (2020). Short-term vegetation loss versus decadal degradation of grasslands in the Caucasus based on Cumulative Endmember Fractions. *Remote Sensing of Environment*, *248*, Article 111969.

Mammides, C. (2020). A global analysis of the drivers of human pressure within protected areas at the national level. *Sustainability Science*, *15*, 1223–1232.

Mammides, C., Zotos, S., & Martini, F. (2024). Quantifying the amount of land lost to artificial surfaces in European habitats: A comparison inside and outside Natura 2000 sites using a quasi-experimental design. *Biological Conservation*, *293*, Article 110556.

Maxwell, S. L., Cazalis, V., Dudley, N., Hoffmann, M., Rodrigues, A. S. L., Stolton, S., Visconti, P., Woodley, S., Kingston, N., Lewis, E., Maron, M., Strassburg, B. B. N., Wenger, A., Jonas, H. D., Venter, O., & Watson, J. E. M. (2020). Area-based conservation in the twenty-first century. *Nature*, *586*, 217–227.

Montalvo Mancheno, C. S., Zazanashvili, N., & Beruchashvili, G. (2016). Effectiveness of the network of protected areas of the South Caucasus at representing terrestrial ecosystems after the dissolution of the Soviet Union. *Environmental Conservation*, *44*, 158–165.

Mysterud, A. (2006). The concept of overgrazing and its role in management of large herbivores. *Wildlife Biology*, *12*, 129–141.

Neal, T. (2024). Estimating the effectiveness of forest protection using regression discontinuity. *Journal of Environmental Economics and Management*, *127*, Article 103021.

Nelson, A., & Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: A global analysis using matching methods. *PLoS ONE*, *6*, Article e22722.

Neudert, R. (2015). Is individualized rangeland lease institutionally incompatible with mobile pastoralism? - A case study from post-socialist Azerbaijan. *Human Ecology*, *43*, 785–798.

Neudert, R. (2021). Combating pasture degradation in Central Asia and the Caucasus—A review of approaches. In V. Beckmann (Ed.), *Transitioning to sustainable life on land* (pp. 231–273). MDPI books.

Neudert, R., Etzold, J., Münzner, F., Manthey, M., & Busse, S. (2013). The opportunity costs of conserving pasture resources for mobile pastoralists in the Greater Caucasus. *Landscape Research*, *38*, 499–522.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., & Policansky, D. (1999). Revisiting the commons: Local lessons, global challenges. *Science*, *284*, 278–282.

Piipponen, J., Jalava, M., de Leeuw, J., Rizayeva, A., Godde, C., Cramer, G., Herrero, M., & Kummu, M. (2022). Global trends in grassland carrying capacity and relative stocking density of livestock. *Global Change Biology*, *28*, 3902–3919.

Pozo, R. A., Cusack, J. J., Acebes, P., Malo, J. E., Traba, J., Iranzo, E. C., Morris-Trainor, Z., Minderman, J., Bunnefeld, N., Radic-Schilling, S., Moraga, C. A., Arriagada, R., & Corti, P. (2021). Reconciling livestock production and wild herbivore conservation: Challenges and opportunities. *Trends in Ecology & Evolution*, *36*, 750–761.

Pressey, R. L., Visconti, P., & Ferraro, P. J. (2015). Making parks make a difference: Poor alignment of policy, planning and management with protected-area impact, and ways forward. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*, Article 20140280.

Pressey, R. L., Visconti, P., McKinnon, M. C., Gurney, G. G., Barnes, M. D., Glew, L., & Maron, M. (2021). The mismeasure of conservation. *Trends in Ecology & Evolution*, *36*, 808–821.

Radeloff, V. C., Beaudry, F., Brooks, T. M., Butsic, V., Dubinin, M., Kuemmerle, T., & Pidgeon, A. M. (2013). Hot moments for biodiversity conservation. *Conservation Letters*, *6*, 58–65.

Radvanyi, J., & Muduyev, S. S. (2007). Challenges facing the mountain peoples of the Caucasus. *Eurasian Geography and Economics*, *48*, 157–177.

Rodrigues, A. S. L., & Cazalis, V. (2020). The multifaceted challenge of evaluating protected area effectiveness. *Nature Communications*, *11*, Article 5147.

Sappington, J. M., Longshore, K. M., & Thompson, D. B. (2007). Quantifying landscape ruggedness for animal habitat analysis: A case study using bighorn sheep in the Mojave Desert. *The Journal of Wildlife Management*, *71*, 1419–1426.

Schleicher, J., Peres, C. A., Amano, T., Llactayo, W., & Leader-Williams, N. (2017). Conservation performance of different conservation governance regimes in the Peruvian Amazon. *Scientific Reports*, *7*, Article 11318.

Schulze, K., Knights, K., Coad, L., Geldmann, J., Leverington, F., Eassom, A., Marr, M., Butchart, S. H., Hockings, M., & Burgess, N. D. (2018). An assessment of threats to terrestrial protected areas. *Conservation Letters*, *11*(3), Article e12435.

Smith, W. K., Dannenberg, M. P., Yan, D., Herrmann, S., Barnes, M. L., Barron-Gafford, G. A., Biederman, J. A., Ferrenberg, S., Fox, A. M., Hudson, A., Knowles, J. F., Macbean, N., Moore, D. J. P., Nagler, P. L., Reed, S. C., Rutherford, W. A., Scott, R. L., Wang, X., & Yang, J. (2019). Remote sensing of dryland ecosystem structure and function: Progress, challenges, and opportunities. *Remote Sensing of Environment*, *233*, Article 111401.

Song, X.-P., Hansen, M. C., Stehman, S. V., Potapov, P. V., Tyukavina, A., Vermote, E. F., & Townshend, J. R. (2018). Global land change from 1982 to 2016. *Nature*, *560*, 639–643.

Soofi, M., Ghoddousi, A., Zeppenfeld, T., Shokri, S., Soufi, M., Jafari, A., Ahmadpour, M., Qashqaei, A. T., Egli, L., Ghadirian, T., Chahartaghi, N. R., Zehzad, B., Kiabi, B. H., Khorozyan, I., Balkenhol, N., & Waltert, M. (2018). Livestock grazing in protected areas and its effects on large mammals in the Hyrcanian forest, Iran. *Biological Conservation*, *217*, 377–382.

Stanimirova, R., Arévalo, P., Kaufmann, R. K., Maus, V., Lesiv, M., Havlík, P., & Friedl, M. A. (2019). Sensitivity of global pasturelands to climate variation. *Earth's Future*, *7*, 1353–1366.

Strömberg, C. A. E., & Staver, A. C. (2022). The history and challenge of grassy biomes. *Science*, *377*, 592–593.

Vicente-Serrano, S. M., & National Center for Atmospheric Research Staff. (2010). *The climate data guide: Standardized Precipitation Evapotranspiration Index (SPEI)*. https://climatedataguide.ucar.edu/climate-data/standardized-precipitation-evapotranspiration-index-spei

Vogt, J. V., Safriel, U., Von Maltitz, G., Sokona, Y., Zougmore, R., Bastin, G., & Hill, J. (2011). Monitoring and assessment of land degradation and desertification: Towards new conceptual and integrated approaches. *Land Degradation & Development*, *22*, 150–165.

Watson, J. E., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, *515*, 67–73.

Wiesmair, M., Feilhauer, H., Magiera, A., Otte, A., & Waldhardt, R. (2016). Estimating vegetation cover from high-resolution satellite data to assess grassland degradation in the Georgian Caucasus. *Mountain Research and Development*, *36*(1), 56–65.

Wolf, C., Levi, T., Ripple, W. J., Zárrate-Charry, D. A., & Betts, M. G. (2021). A forest loss report card for the world's protected areas. *Nature Ecology & Evolution*, *5*, 520–529.

Wooldridge, J. (2011). *Introductory econometrics: A modern approach*. Cengage South-Western, Mason, Ohion.

Zazanashvili, N., Sanadiradze, G., Garforth, M., Bitsadze, M., Manvelyan, K., Askerov, E., Mousavi, M., Krever, V., Shmunk, V., Kalem, S., & Devranoğlu Tavsel, S. (Eds.). (2020). *Ecoregional Conservation Plan for the Caucasus: 2020 Edition*. Tbilisi.

Zhao, H., Wu, R., Long, Y., Hu, J., Yang, F., Jin, T., Wang, J., Hu, P., Wu, W., Diao, Y., & Guo, Y. (2019). Individual-level performance of nature reserves in forest protection and the effects of management level and establishment age. *Biological Conservation*, *233*, 23–30.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

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