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Establishing baselines for trophic rewilding: mammal density, occupancy, and activity patterns in two Italian sites

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Title

Establishing baselines for trophic rewilding: mammal density, occupancy, and activity patterns in two Italian sites

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Abstract

Trophic rewilding has emerged as a promising approach to restore ecological processes through the reintroduction of species. Yet, rigorous pre-intervention assessments of native wildlife communities remain scarce, limiting the ability to detect and interpret rewilding outcomes. Here, we provide a baseline assessment of wild mammal populations in two sites of central Italy where trophic rewilding plans are being proposed: the Alfina plateau (Umbria) and the Diaccialone estate (Tuscany). We focused on six target species: European badger, martens (*Martes spp.*), porcupine, red fox, roe deer, and wild boar. By using a standardized camera-trapping design we estimated mammal population density (using Random Encounter Models), modelled species occupancy and its environmental drivers, and quantified diel activity patterns and interspecific temporal overlap. Density estimates varied considerably between species and sites, with roe deer and wild boar showing the highest densities in both areas. Occupancy models revealed high species presence across both sites, with detectability and space use shaped by landscape structural complexity and anthropogenic disturbance. Activity analyses highlighted interspecific differences in temporal niche partitioning and some significant inter-site variation. Together, these results establish a robust baseline of the native mammal communities prior to the planned rewilding interventions, allowing to inform rewilding strategies and plan before-after comparisons. Based on our findings, we provide a set of practical recommendations for future rewilding plans.

Introduction

Accelerating anthropogenic pressures are driving global biodiversity loss and eroding ecosystem functions (Wang et al. 2025). In response to this crisis, international frameworks such as the Kunming–Montreal Global Biodiversity Framework (CBD 2022) and the EU Nature Restoration Law (EC 2024) call for large-scale ecosystem recovery, highlighting the need for integrated conservation strategies which include ecological restoration and rewilding (Mutillod et al. 2024). Rewilding is defined as the process of recovery of degraded ecosystems by restoring natural processes and the food web at all trophic levels to achieve self-sustaining and resilient communities (Perino et al. 2019; Carver et al. 2021). Two main approaches exist: passive rewilding, which relies on natural succession following land abandonment, and trophic rewilding, which involves reintroducing species to restore lost trophic interactions and ecosystem functions (Svenning et al. 2016; Perino et al. 2019). Most trophic rewilding initiatives focus on re-establishing large herbivore populations, which can be crucial in sustaining ecosystems and biodiversity, but have been disproportionately extirpated in the Anthropocene (Svenning et al. 2019).

Despite being proposed decades ago, rewilding remains highly debated among conservationists due to ambiguous definitions, limited empirical evidence supporting many initiatives, and concerns about ecological predictability, socio-economic trade-offs, and ethical implications of species reintroductions (Nogués-Bravo et al. 2016; Hayward et al. 2019; Hoeks et al. 2023). Moreover, with few noticeable exceptions (e.g., Tanentzap et al., 2023), most rewilding projects started without baseline assessments and proper monitoring strategies in place, hindering the ability to assess their effective impact on biodiversity and ecosystem functions (Segar et al. 2022). Robust assessment of rewilding outcomes requires integrated monitoring plans including biodiversity, ecosystem services, and socio-economic aspects, conducted

across multiple time periods to allow meaningful comparisons. Several protocols have been recently proposed to track rewilding progress on large spatio-temporal scales (Torres et al. 2018; Perino et al. 2019; Segar et al. 2022). While these frameworks help standardize monitoring, they mostly rely on expert-based assessments. Thus, there is still a need for analytical, data-driven, approaches to empirically inform rewilding plans (Hart et al. 2023).

Establishing ecological baselines prior to trophic rewilding is essential for quantifying the structure and functioning of native communities and for predicting potential cascading responses to the intervention. Baseline density assessments of native wildlife allow detection of changes in population abundance and demographic trajectories through time, providing a quantitative foundation for assessing potential competitive release, facilitation, or displacement following rewilding (Callaghan et al. 2024). Similarly, occupancy modelling provides robust estimates of species spatial distribution while explicitly accounting for imperfect detection, allowing the identification of areas where rewilding interventions may differentially influence wildlife presence or habitat use (MacKenzie et al. 2017). Finally, assessing activity patterns, in terms of diel behaviour and temporal niche partitioning, allows evaluating wildlife behavioural responses to changes in disturbance regimes and interspecific interactions, potentially highlighting early signals that may precede detectable demographic shifts (Ridout & Linkie 2009a). Collectively, these metrics can provide a rigorous baseline against which ecological changes can be quantified, offering a robust foundation for monitoring the outcomes and effectiveness of trophic rewilding interventions. Camera trapping is an efficient, scalable, and cost-effective tool to derive multiple wildlife population metrics (including density, occupancy, activity) through a standardized and non-invasive method that can operate continuously across diverse environmental conditions and for a wide range of species (Burton et al. 2015).

Two trophic rewilding initiatives are currently being evaluated in central Italy by Fondazione Riselva, a private foundation. The initiative focuses on two target areas: the Alfina Plateau, a 1500 ha agricultural landscape in the Umbria region, and the Diaccialone estate, an 800 ha area in Tuscany characterized by the presence of semi-free-ranging livestock. Planned interventions include the reintroduction of wild and semi-wild large herbivores to restore lost trophic interactions and foster self-regulating ecosystems, while simultaneously promoting a nature-based economy in currently underused rural landscapes. Here, we provide an integrated baseline assessment of medium- and large-sized mammal communities across the two target sites by estimating species density, occupancy, and activity patterns using a standardized camera-trapping design. This study has two main objectives. First, by characterizing wild mammal communities, we aim to derive planning recommendations for the upcoming reintroductions, including their potential implications on resident wildlife. Second, we aim to establish a rigorous pre-intervention baseline, providing comparable, data-driven metrics of wildlife populations that will support the monitoring of ecological responses following rewilding. This work is part of a broader effort to establish comprehensive pre-rewilding baselines across multiple taxonomic groups, including small mammals, bats, birds, insects, and plants.

Material and methods

Study area

The Alfina plateau is located in central Italy, at the border between Umbria (province of Terni) and Lazio (province of Viterbo; Fig. 1a). The study area we used for the samplings covers around 1500 ha, and is situated at an average distance of 3.98 km north of Lake Bolsena and at a mean altitude of 612 m above sea level. The area is dominated by open vegetation, mainly improved meadows/pastures and monospecific crops, both subject to intense disturbance and low naturalness. The remaining is covered by *Quercus cerris*-dominated woodlands, partly accompanied by *Corylus avellana* and *Castanea sativa*, with a minor presence of *Pinus nigra* plantations currently undergoing spontaneous recolonization.

The Diaccialone estate is a 600 ha private property located in Tuscany (province of Grosseto), approximately 13 km from the Tyrrhenian coast and at a mean altitude of 200 m above sea level (Fig. 1b). The area is predominantly wooded, with deciduous woodlands dominated by *Quercus cerris* and *Quercus frainetto*, and mixed thermophilous woodlands of *Fraxinus ornus*, *Acer monspessulanum*, and *Quercus ilex*. The remaining is covered by open vegetation, including monospecific crops and semi-natural grasslands retaining a relatively good degree of naturalness. The estate hosts semi-free-ranging herds of Maremma cattle (*Bos taurus*) and fenced buffaloes (*Bubalus bubalis*), managed within the property. In addition, we chose to include part of the Poggi Alti estate, which borders Diaccialone on the western side, to increase sampling effort and spatial replication, thereby improving the robustness of the analysis. The Poggi Alti estate covers over 500 ha, mostly consisting of forested areas and includes an enclosed fenced area where animals are introduced for hunting purposes. For our sampling, we

used the northern part of the area, located outside the fence, which covers ca. 200 ha for a total sampled area of ca. 800 ha.

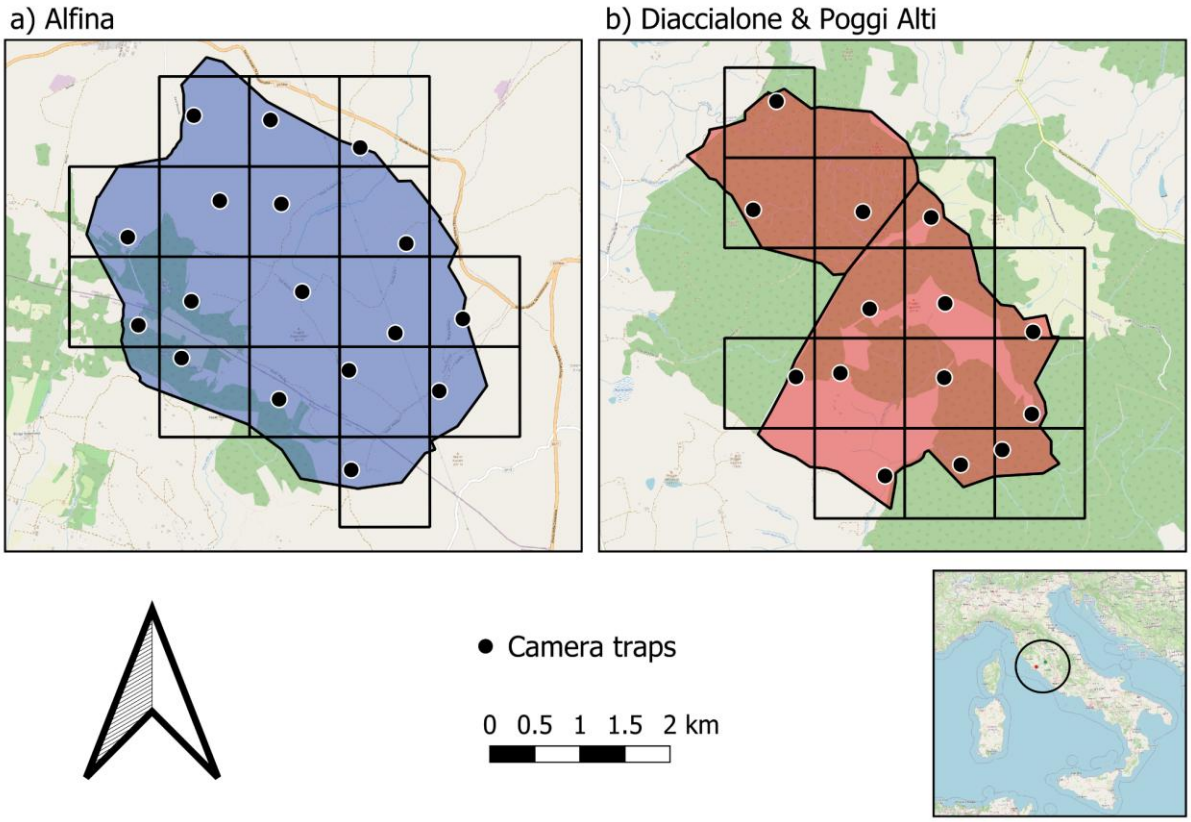


Figure 1. *Map of the two study areas showing the eastern part of a) the Alfina Plateau (blue area) and b) the Diaccialone estate, including the northern portion of Poggi Alti (red area). A grid of 1 km² quadrants was used for camera trap deployment, with black dots indicating camera locations.*

Data collection

We conducted camera trapping surveys in the study areas between 2024 and 2025 to estimate the population density, occupancy, and activity patterns of medium- and large-sized mammal species. In the Alfina plateau, sampling was carried out in 2024 using 17 camera traps from the 10th of April to the 27th of July, while in the Diaccialone estate sampling was conducted in

2025 using 14 camera traps from the 7th of April to the 17th of July. Camera placement followed a systematic random sampling design based on a 1 km × 1 km grid, with one camera deployed within each grid cell using the cell centroid as a reference point (Fig. 1).

All camera traps (Browning Dark Ops Pro DCL Nano) were set in photo mode with a capture delay of 1 second, three rapid-fire shots per trigger, and an image resolution of 1280 × 720 (HD). Cameras were mounted on trees at heights ranging from 0.7 to 1.1 m and oriented northward to minimize direct exposure to sunlight. Where vegetation cover or terrain constraints prevented northward orientation, the closest feasible orientation was selected while ensuring to avoid direct sunlight. Camera traps were checked approximately every three weeks to replace batteries and memory cards. No baits were used to attract wildlife.

We focused our analyses on meso- and macro-mammals: European badger (*Meles meles*), pine (*Martes martes*), beech martens (*M. foina*), porcupine (*Hystrix cristata*), red fox (*Vulpes vulpes*), roe deer (*Capreolus capreolus*), and wild boar (*Sus scrofa*). Pine marten and stone marten observations were grouped as *Martes spp.*, as the two species are often indistinguishable from camera trap images alone without the support of additional evidence such as carcass examination or track analysis, particularly for nocturnal photos taken with infrared (IR) rather than white/incandescent flash (Manzo et al. 2012). Wolf (*Canis lupus italicus*) presence was also recorded in both study areas (a breeding pair in Alfina and a stable pack in Diaccialone). However, wolf observations were not included in the analyses because the home range of a wolf pack is much larger than our study areas, and reliable estimates of wolf population parameters would require surveys conducted at a broader spatial scale (Mancinelli et al. 2018).

Population density

We used Random Encounter Models (REM; Rowcliffe et al. 2008) to estimate the population densities of the target mammal species in the two areas. REM treats individual animals like ideal gas particles, and estimates density based on the random encounters between animals and camera traps within the collective detection viewsheds of a camera array. To ensure reliable estimates, population densities were estimated only for species with a minimum of 40 encounters (Palencia & Barroso 2024). REM uses the following formula to estimate density (D):

$$D = \frac{y}{t} \frac{\pi}{vr(2+a)} \quad [1]$$

where *y* is the number of encounters, *t* is total camera survey effort (in days), *v* is the average distance traveled by an individual during a day (i.e., day range, expressed in km/day), and *r* and *a* are respectively the radius (meters) and angle (degrees) of the camera traps detection zone (Rowcliffe et al. 2008). To estimate day range, radius, and angle, we performed a subsampling of images based on a 30-minute independence threshold. To estimate encounter rate, we considered each time that an individual of the target species entered the detection zone of the camera trap as a new encounter. The parameters required for the REM (i.e., animal speed, radial distance, and angle from the camera) were extracted from camera trap data through the Agouti platform, a standardised tool for camera trap data management and analysis (ENETWILD-Consortium et al. 2022). Camera calibration was performed following the photogrammetric protocol provided by the European Observatory of Wildlife (EOW; www.wildlifeobservatory.org), which allows the estimation of real distances and angles from images using reference objects of known size placed in the field of view. To account for animal curiosity towards the camera trap, which can bias speed and distance estimates, we excluded

from the analysis those encounters in which individuals appeared to interact with the device, retaining only records representative of natural movement behaviour (Bessone et al. 2020). When a group of individuals was observed in the same frame, the distance travelled was measured based on the first individual to enter the field of view, while the distance and angle from the camera were measured based on the individual closest to the centre of the field of view (Rowcliffe et al. 2016; Dragonetti et al. 2025). Day range was estimated using the "activity" and "trappingmotion" packages in R (Rowcliffe 2016; Palencia et al. 2021). Speed was obtained by identifying different movement behaviours through *k*-means cluster analysis, and estimating the average speed for each behaviour as its harmonic mean (Palencia et al. 2021), while activity level was estimated as the frequency of encounters throughout the day, with each encounter replicated according to the number of individuals detected (e.g., an encounter of four individuals was counted four times; Rowcliffe et al. 2014). Finally, day range was estimated as the sum of the product of the mean speed and the activity level associated with each behaviour (Palencia et al. 2021). To estimate the detection zone, we recorded the radial distance and angle of each animal when it first triggered the camera trap and applied a distance sampling analysis to estimate the effective detection radius and angle (Buckland et al. 2001; Rowcliffe et al. 2011). Additional details on the detection zone estimation are provided in the Supporting materials (Text S1).

Occupancy

Occupancy ($\boldsymbol{\Psi}$) can be defined as the proportion of an area occupied by a species, or fraction of landscape units where the species is present (i.e., the site contains at least one individual of the species; MacKenzie et al. 2002). However, since in natural environments the probability of detecting a species is always less than one, occupancy ($\widehat{\Psi}$) can be estimated as:

$$\widehat{\Psi} = \frac{\hat{x}}{s} \quad [2]$$

where s is the number of total sites and $\hat{x}$ is the estimated number of occupied sites, derived from the number of sites (n) in which a species was detected and the probability of detecting that species ($\hat{p}$) (MacKenzie et al. 2002):

$$\hat{x} = \frac{n}{\hat{p}} \quad [3]$$

We modelled single-season, single-species, occupancy models to estimate the occupancy of the target mammal species in the two areas. The initial set of environmental covariates considered for occupancy modelling included variables describing primary productivity, land-cover composition, habitat configuration, and anthropogenic disturbance. Primary productivity was quantified using Normalized Difference Vegetation Index (NDVI), employed as a proxy for vegetation vigour and biomass. Land-cover composition was represented by the percentage cover of "Tree cover", "Grassland", "Cropland", and "Built-up" classes within each sampling site. Habitat fragmentation was characterised using landscape-level metrics computed within the 1 km² grid cell containing each camera trap location: Fractal Dimension Index, measuring patch shape complexity; Mean Patch Shape Ratio, reflecting perimeter-to-area relationships; Overall Core Area, representing the amount of core habitat unaffected by edge effects (edge being defined as the outermost pixels of each patch); Mean Patch Area, indicating average patch size; and Largest Patch Index, expressing the proportion of the cell occupied by the largest patch. Finally, anthropogenic influence and landscape context were captured through minimum Euclidean distance to roads and livestock presence, used as proxies for disturbance intensity. Additional details on the covariates extraction are provided in the Supporting materials (Text S2).

Before running the models, we standardized the covariates (centered to zero and scaled based on unit variance) and assessed collinearity among them using Spearman's correlation

coefficient (ρ), retaining only pairs with $|\rho| < 0.7$. NDVI, land-cover, and anthropogenic disturbance variables were included as candidate covariates for both occupancy (ψ) and detection (p), based on the assumption that these factors can influence species space use as well as the probability of detecting individuals with camera traps (e.g., by shaping vegetation structure and visibility around the sensor, or by affecting animal movement near camera locations, Kolowski & Forrester 2017; Hofmeester et al. 2019). In contrast, fragmentation metrics were only included as candidate covariates for occupancy (ψ), as we lacked a clear ecological rationale for a direct effect of landscape configuration complexity on detection probability.

To avoid underestimating the standard errors, following the goodness-of-fit test on each species' global model (MacKenzie & Bailey 2004), we identified overdispersed models ($\hat{c} > 1$, p-value < 0.05) and used $\hat{c}$-adjusted standard errors (SEadj) for those species/models:

$$SEadj = SE \times \sqrt{\hat{c}} \quad [4]$$

For each species, we conducted model selection for both detection (p) and occupancy (ψ) covariates simultaneously, considering all possible combinations up to a maximum of 2 degrees of freedom (i.e., up to 2 variables in any combination of p and ψ). This limit was necessary to avoid model overfitting given the limited number of sites per area, while we prefer to not perform a stepwise selection as this could introduce selection bias when the optimal detection structure is conditional on the occupancy structure, and vice versa (Doherty et al. 2012). Model ranking was based on AICc (Akaike Information Criterion corrected for small sample sizes), which reduces the risk of overfitting associated with AIC in small samples (Sugiura 1978; Hurvich & Tsai 1989). Model averaging was then performed on converged models only (i.e., excluding those with occupancy parameter estimates > 5 or standard errors > 10 in the logit scale; Steen et al. 2023), using Akaike weights, derived from each model's AICc, as averaging weights. Finally, the relative variable importance (RVI) of each predictor was calculated as the

sum of the Akaike weights of all models in which it occurred, to identify the most explanatory variables for each species (Burnham & Anderson 2002). Based on the resulting averaged model, we predicted the occupancy and mean detection probability of each species in each grid cell of the study area. All occupancy models were built using the R package "unmarked" (Fiske & Chandler 2011).

Activity patterns

We quantified activity patterns of mammal species using time-stamped camera-trap detections, treating time of detection as a circular variable over the 24-hour cycle (Rowcliffe et al. 2014). Detection times were converted into radians and analysed using kernel density estimation to derive continuous diel activity curves and to estimate overall activity levels. Each detection event was replicated according to the observed number of individuals in each separate encounter. To quantify the degree of temporal overlap in species activity patterns, we calculated the overlap coefficient (Δ) between pairs of diel activity curves using the R package "overlap" (Ridout & Linkie 2009). The coefficient ranges from 0 (no temporal overlap) to 1 (complete overlap) and represents the area under the curve shared by two density distributions, defined as:

$$\Delta = \int_0^{2\pi} \min\left(f(\theta), g(\theta)\right) d\theta \quad [5]$$

where $f(\theta)$ and $g(\theta)$ are the kernel density functions of the two activity distributions estimated on the circular scale $[0, 2\pi]$.

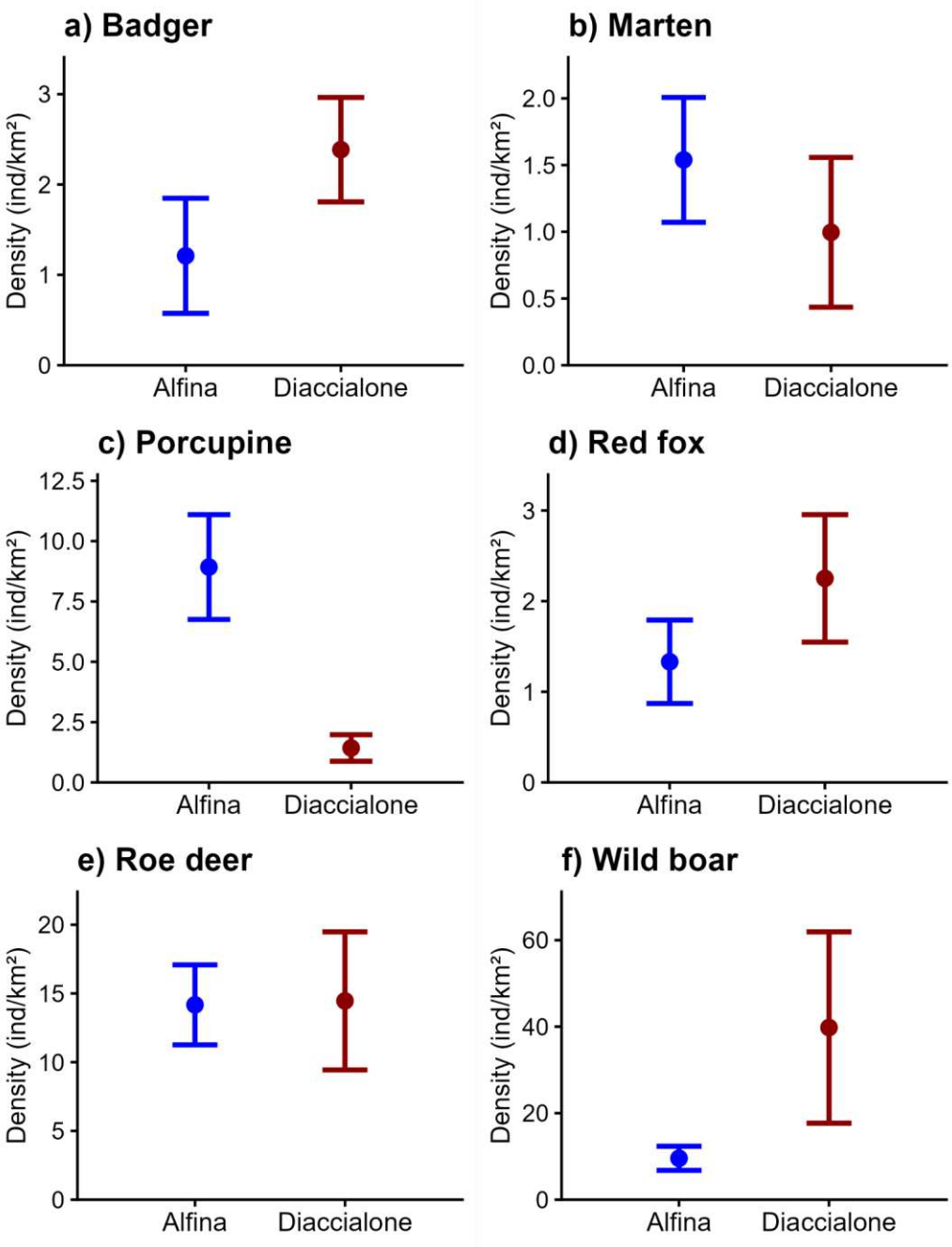
We used the Dhat4 estimator, recommended for sample sizes ≥ 50 (Ridout & Linkie 2009). Confidence intervals (95%) were obtained via percentile bootstrap resampling with 1000

iterations. First, we performed pairwise comparisons among all six target species within each study area, to assess interspecific temporal partitioning. Then, for each species separately, we compared activity patterns between the two study areas to evaluate intraspecific temporal consistency and differences across sites. The statistical significance of differences in overall activity levels between pairs of curves was assessed using a Wald test implemented in the "activity" package (Rowcliffe 2016), with $p < 0.05$ considered statistically significant.

Results

Population density

Species population densities varied between the two study areas (Fig. 2, Tab. S1, S2). The Alfina area hosted much higher density of porcupine compared to Diaccialone, as well as higher density of marten species. The Diaccialone area hosted much higher density of wild boar compared to Alfina (even accounting for the high estimate uncertainty), as well as higher densities of European badger and red fox. Roe deer density was equivalent in the two areas. Regarding the REMs intermediate results, Alfina had higher activity levels and higher day range for the red fox and the wild boar compared to Diaccialone, as well as higher day range for marten species (Tab. S1, S2). Diaccialone had higher activity level for marten species compared to Diaccialone and much higher day range for the porcupine (Tab. S1, S2).



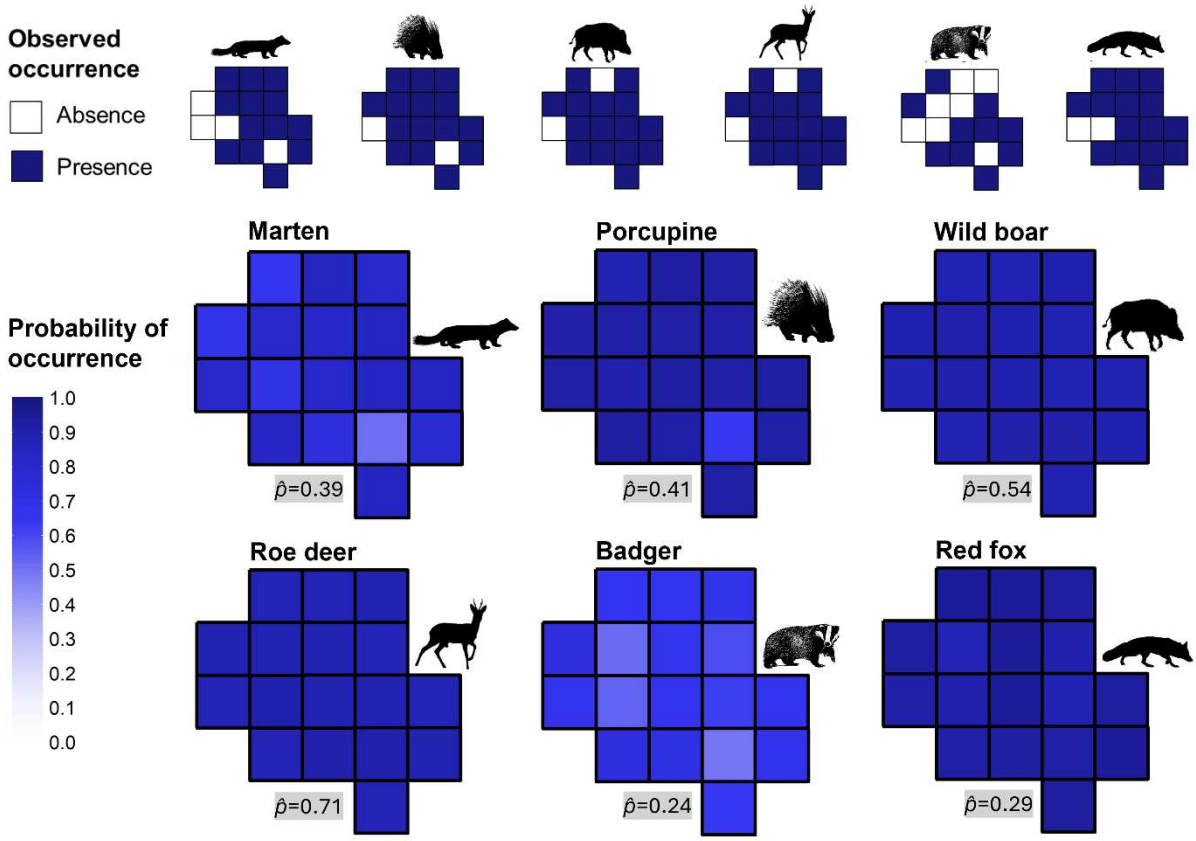
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295 ***Figure 2.*** *Mammal species densities in the two areas (dots) and their confidence intervals*
296 *(whiskers), estimated in 2024 (Alfina) and 2025 (Diaccialone), by using the Random Encounter*
297 *model (REM). Marten density is referred to both pine marten (*Martes martes*) and beech*
298 *marten (*Martes foina*), since it was not possible to distinguish the two species in most of the*
299 *pictures.*

Occupancy

In both sampling areas we recorded high values of naïve occupancy for all species (Naïve Ψ between 0.58 and 1, Tab. S3). However, the mean detectability estimates ($\hat{p}$) showed a substantial difference between the two areas, with higher estimated values for all species in the Alfina area compared to Diaccialone ($\hat{p}$ ranging from 0.05-0.14 in Diaccialone, 0.24-0.71 in Alfina; Fig. 3, 4, Tab. S3).

Due to the high naïve occupancy values of the species, model-averaged occupancy coefficients were generally associated with large standard errors, and no occupancy covariate achieved an RVI greater than 0.8 in either study area, indicating considerable uncertainty in the effects of individual predictors (Tab. S4, S5). Despite not having strong statistical support, in Diaccialone a negative relationship was observed between the presence of *Martes* spp. and the proximity to grazing areas, whereas in Alfina these species show a tendency to avoid grasslands.



313

314 ***Figure 3.*** *Naïve occupancy and predicted occurrence estimates with corresponding standard*
315 *errors of the target species in the Alfina plateau. Detectability estimates (p̂) refer to seven-day*
316 *sampling occasions (i.e., the average probability of observing the species at a random site*
317 *when present during a sampling week). Marten results refer to both pine marten (*Martes
318 martes*) and beech marten (*Martes foina*), since it was not possible to distinguish the two*
319 *species in most of the camera-trap pictures.*

320

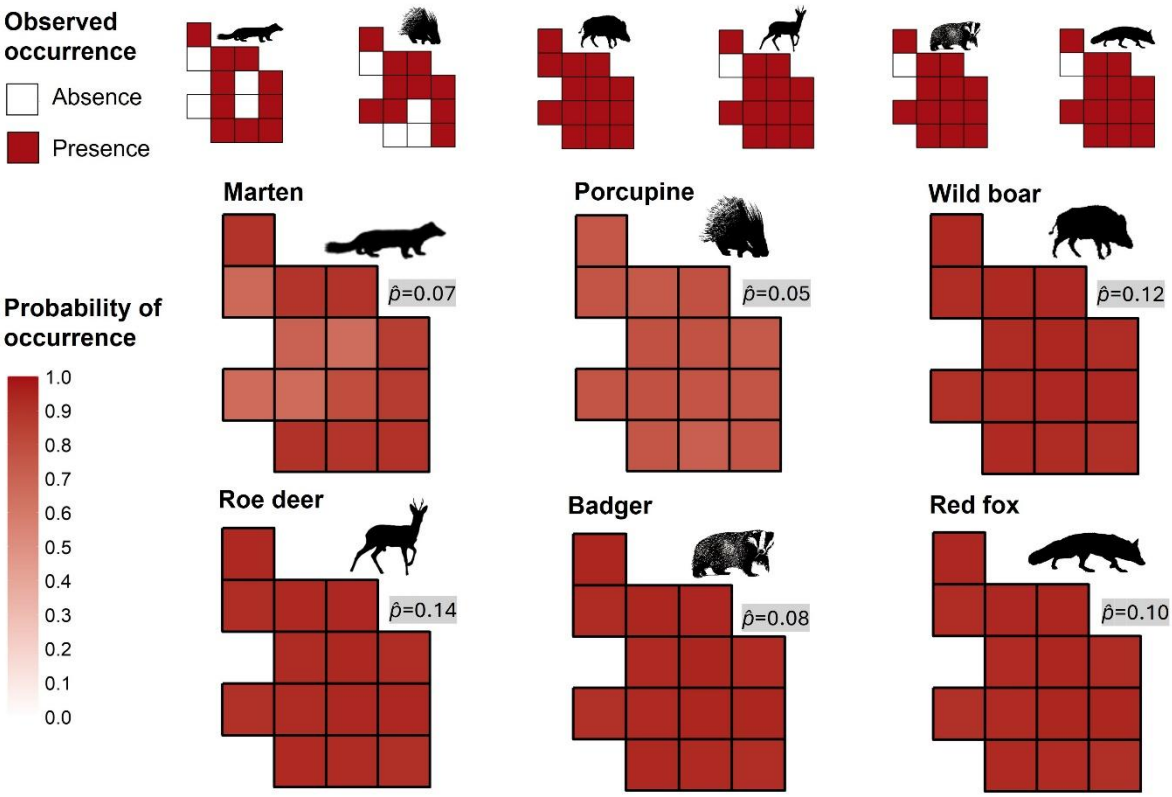


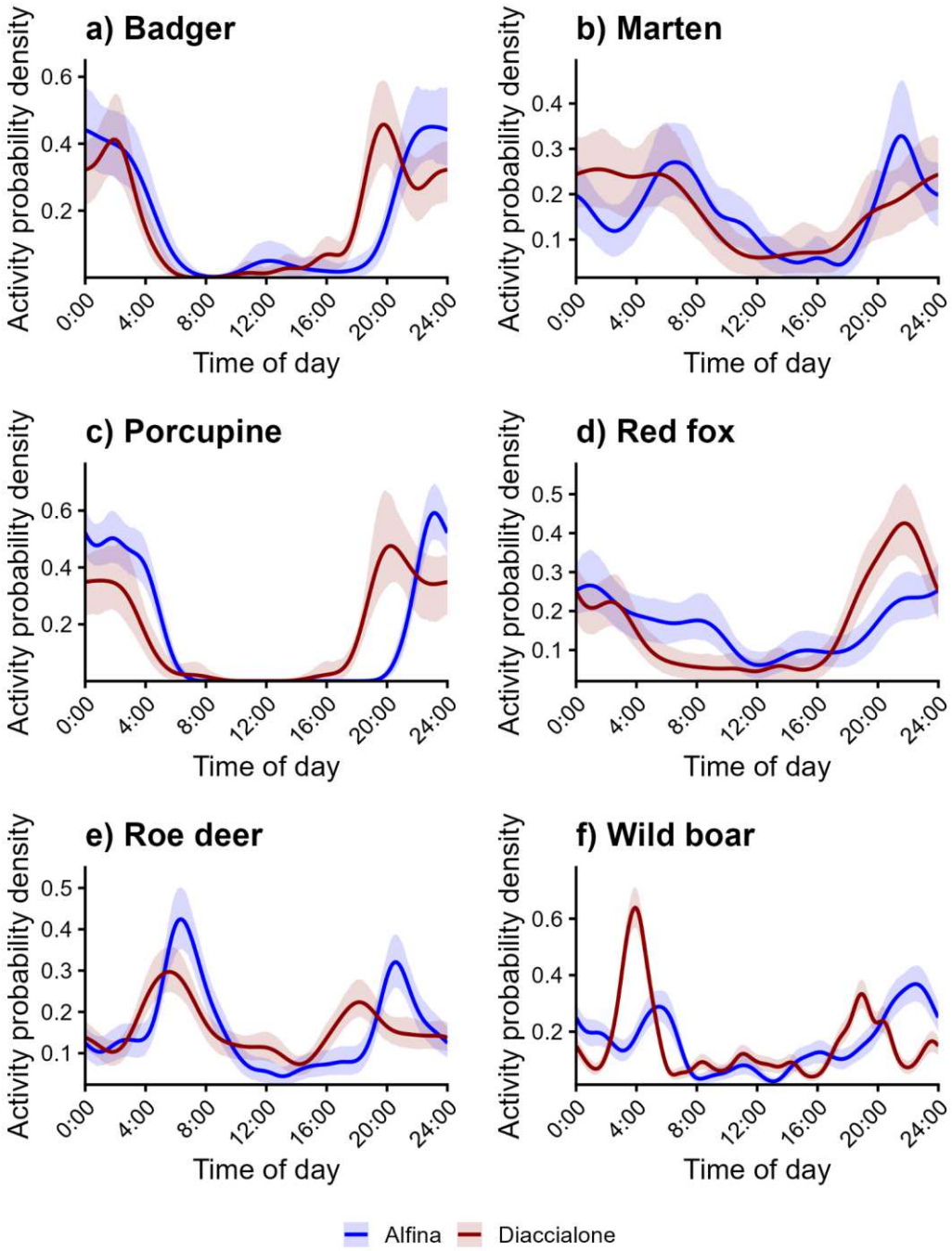
Figure 4. *Naïve occupancy and predicted occurrence estimates with corresponding standard errors of the target species in the Diaccialone estate. Detectability estimates (p̂) refer to seven-day sampling occasions (i.e., the average probability of observing the species at a random site when present during a sampling week). Marten results are referred to both pine marten (*Martes martes*) and beech marten (*Martes foina*), since it was not possible to distinguish the two species in most of the camera-trap pictures.*

Activity patterns

Activity patterns showed both similarities and differences between the two study areas (Fig. 4; Tab. S6, S7). The European badger and porcupine showed marked nocturnal activity in both areas. In contrast, marten species displayed extended activity across the full 24-hour cycle in both areas. The red fox showed the clearest inter-area difference, with a longer and more evenly

distributed daily activity in Alfina (14.35 ± 1.61 h) compared to a shorter, more strictly nocturnal pattern in Diaccialone (9.00 ± 0.94 h). The roe deer showed consistent bimodal, crepuscular activity in both areas, while wild boar activity was predominantly nocturnal at both sites but substantially longer in Alfina (10.37 ± 0.89 h) than in Diaccialone (5.86 ± 0.34 h).

Looking at interspecific activity overlap (Tab. S8, S9), porcupine showed the most pronounced temporal segregation from the other target species, particularly from the roe deer, consistent with its strictly nocturnal habits contrasting with the crepuscular pattern of the latter. Badger showed moderate overlap with several species, most notably red fox in Alfina and marten in Diaccialone. Red fox, roe deer, and wild boar instead showed consistently high mutual overlap in both areas, suggesting broadly similar diel activity patterns among these species.



348

349 ***Figure 5.*** *Daily activity patterns of the six target species at the Alfina plateau (2024) and the*
350 *Diaccialone estate (2025), estimated using circular kernel density functions based on*
351 *camera-trap detections. Solid lines represent the estimated activity probability density, while*
352 *shaded areas indicate 95% confidence intervals. Full statistical details of the overlap*
353 *coefficient are available from Table S7. Marten activity refers to pooled detections of* Martes

spp.*, as pine marten (*M. martes*) and stone marten (*M. foina*) could not be reliably distinguished in most camera-trap images.*

Discussion

By using standardised camera trap surveys, we were able to obtain key demographic and ecological parameters of several mammal species in two areas subject to future rewilding plans. These metrics capture different but interrelated ecological dimensions and are particularly well suited to detect both demographic and behavioural responses to the planned reintroduction of wild and semi-wild large herbivores. We found density estimates for the species were largely in line with values reported for other European and Italian populations, with some noticeable difference between the two study areas. Densities of the European badger and red fox were comparable between sites, and in line with those previously estimated using camera traps in semi-natural and rural environments in Italy (Balestrieri et al. 2016; Zampetti et al. 2024; Dragonetti et al. 2025) and Europe (Guerrasio et al. 2024). The density of martens was higher in Alfina than in Diaccialone, and exceeded values reported in central Italy (Manzo et al. 2012), likely reflecting our pooling of *M. martes* and *M. foina* observations. Porcupine density was markedly higher in Alfina than in Diaccialone and exceeded most published estimates from Italy (Franchini et al. 2022; Palencia et al. 2024; Dragonetti et al. 2025), suggesting that the Alfina plateau may represent an area of particularly high suitability for the species, characterized by extensive agricultural and semi-natural habitats. Roe deer densities were similarly high in both areas and fell within the medium–high European range (Melis et al. 2009; Guerrasio et al. 2024), indicating broadly comparable habitat suitability between the two sites and conditions favorable to the species. In contrast, wild boar density was substantially higher

in Diaccialone than Alfina, yet remained within the range reported for other high-density wild boar populations in Italy and Europe (Fattorini & Ferretti 2020; ENETWILD-consortium et al. 2022; Guerrasio et al. 2024; Dragonetti et al. 2025). This marked difference between the two areas likely reflects the contrasting hunting regimes: while Alfina is subject to hunting pressure, the Diaccialone estate is a private, hunting-free reserve. Such areas can act as refugia for wildlife populations exposed to hunting disturbance in surrounding landscapes, potentially supporting higher local densities (Thurfjell et al. 2013). Nevertheless, the Diaccialone estimate may be inflated, as two camera traps were randomly placed along areas of intense wild boar activity, resulting in particularly high encounter rates.

Occupancy results indicate consistently high presence of all the target species across the study areas, with no clear spatial patterns among grid cells, suggesting that both the Alfina and Diaccialone offer suitable habitat conditions for generalist and adaptable species. Mean detectability estimates were consistently higher in Alfina, likely reflecting differences in habitat structure, as open landscapes tend to facilitate species detectability for camera traps, whereas dense vegetation or heterogeneous habitats, like those in the Diaccialone, can obscure detections (Burton et al. 2015). Human disturbance likely influenced wildlife detectability in Alfina. Specifically, building density decreased the detection probability of roe deer, while proximity to primary roads reduced that of porcupines but increased that of wild boar, consistent with the ecology of these species (Thurfjell et al. 2009; Bonnot et al. 2013; Mori et al. 2022b). Although occupancy covariates showed high standard errors and low variable importance, likely due to the widespread occurrence of species across both study areas, the observed trends may still provide valuable ecological insights. In Diaccialone, most species showed positive associations with shrublands, while in Alfina responses to forest cover and grasslands were more heterogeneous and species-specific. However, all species in both study areas showed positive associations with the fractal dimension index, with the porcupine,

badger, and red fox showing a strong preference for structurally complex habitats with larger ecotonal zones (Wereszczuk & Zalewski 2015; Bischof et al. 2019). Regarding human disturbance, the avoidance of grazing areas by martens is consistent with evidence that livestock can negatively affect space use for certain species through disturbance, vegetation changes, and reduced prey availability (Schieltz & Rubenstein 2016; Pringle et al. 2023). In Alfina, the positive association of martens with human buildings might reflect the dominance of beech marten over pine marten, as the former is more synanthropic. However, this interpretation should be treated with caution, as our *Martes* spp. grouping does not allow us to attribute this pattern to either species individually, and the two species differ substantially in their ecology and habitat use (Wereszczuk & Zalewski 2015).

Activity patterns results differed between the two study areas for some of the target species, likely reflecting the combination of different disturbance regimes, habitat composition, and predator pressure. The nocturnal behaviour of European badger and porcupine is consistent with patterns reported in rural and semi-natural habitats (McClune et al. 2015; Luzi et al. 2021). The similarity in badger mean activity between the areas suggests limited influence of local context on its overall temporal allocation. In contrast, the lower porcupine activity in Alfina, despite higher density estimates, may reflect greater trophic resource availability in this area, where croplands could support abundant and accessible food resources, allowing individuals to meet their energetic requirements in less time compared to Diaccialone. Marten displayed extended activity in both areas, with higher estimates in Diaccialone, possibly reflecting the greater proportion of woodland habitat in that area. The broadly similar patterns between areas suggest limited influence of local conditions on temporal behaviour. However, these results should be interpreted with caution, as partial temporal asynchrony has been reported in pine and stone martens in other studies (Wereszczuk & Zalewski 2015; Monterroso et al. 2016). The differences in red fox mean activity levels between areas are consistent with behavioural

plasticity in response to multiple sources of disturbance (Díaz-Ruiz et al. 2016). Species activity in Diaccialone was more concentrated during nighttime and overall shorter compared to Alfina. This possibly reflects both the daily presence of semi-free-ranging livestock, which generates predictable daytime disturbance, and a more regular wolf presence in the area, which may suppress fox activity through intraguild competition and interference, inducing a more cautious and temporally restricted behavioural pattern (Wikenros et al. 2017). Despite having high temporal overlap between areas, the roe deer showed more pronounced crepuscular peaks in Alfina, likely reflecting stronger temporal constraints imposed by hunting pressure, which could push ungulates towards safer activity windows (Pagon et al. 2013). The higher mean activity observed in Diaccialone may also reflect the need for more extended foraging, where food resources could be more dispersed compared to the mosaic of crops and woodland edges available in Alfina (Krop-Benesch et al. 2013). Wild boar showed the most marked inter-area difference in temporal overlap, with stronger nocturnality and lower mean activity in Diaccialone compared to Alfina. This likely reflects greater dispersion of food resources across the more open landscape of Alfina, requiring more extensive and flexible foraging movements, compared to the denser woodland of Diaccialone where resources are more concentrated (Massei et al. 1996).

Among the most relevant interspecific patterns, the temporal asynchrony of the porcupine compared to several other species (roe deer, red fox, martens) may reflect intrinsically different activity strategies rather than active avoidance (Díaz-Ruiz et al. 2016; Luzi et al. 2021; Mori et al. 2022a). In contrast, the moderate overlap between the badger and red fox in Alfina, and between the badger and marten in Diaccialone, may reflect subtle temporal partitioning driven by interspecific competition, as all three are opportunistic omnivores with partly overlapping dietary niches (Barrull et al. 2014). Finally, the broadly similar diel patterns observed among red fox, roe deer, and wild boar suggest that these species do not partition time as a primary

axis of niche differentiation, relying instead on spatial or trophic differentiation to reduce interspecific competition.

Implications for trophic rewilding plans

Overall, our results provide a baseline for wild mammal presence and activity across both study areas, shaped by a range of environmental and anthropogenic factors discussed above. The introductions of large herbivores are expected to affect landscape complexity (Perino et al. 2019) and resource availability, with potential consequences for the resident mammal community (Greco et al. 2025; Mas-Carrió et al. 2025). For instance, increased grazing and trampling may reduce undergrowth cover, potentially decreasing occupancy of disturbance-sensitive species such as porcupine, while facilitating movement and foraging efficiency for generalist predators such as the red fox (Katona & Coetsee 2019; Kupferschmid et al. 2022). Such ecological processes may also trigger rapid behavioural adjustments across the community: large herbivores can alter space use, activity patterns, and competitive interactions among sympatric species through both habitat modification and resource competition (Mori et al. 2020; Gaudiano et al. 2021). Wild ungulates, such as the roe deer, may alter temporal patterns or space use in response to increased herbivory competition or habitat modification, with potential cascade effects on predators through altered prey availability. Similarly, the extended activity patterns observed in *Martes* spp., particularly in Diaccialone, may shift through changes in prey communities (Pringle et al. 2023).

Based on these considerations, we are able to provide a set of site-specific recommendations for future rewilding plans in Alfina and Diaccialone.

First, wolf presence should be factored into reintroduction planning: their presence was documented in both Alfina and Diaccialone, though presence appears more consistent in Diaccialone, where a stable pack was recorded. In both areas, this raises a risk of targeted predation on reintroduced ungulates, and measures to minimise such risk should be considered, particularly during the calving season. More generally, the conservation and monitoring of predator populations should be treated as an integral component of both rewilding programmes, given their role in regulating herbivore populations and maintaining ecosystem functioning. Second, the relatively ubiquitous presence of wild ungulates should be accounted for to avoid excessive herbivory competition after rewilding. In Diaccialone, wild ungulates are present even in areas already subject to extensive livestock grazing, suggesting that they are already accustomed to coexisting with domestic ones, likely due to the low grazing intensity and temporal segregation. We therefore recommend maintaining low livestock grazing pressure in at least part of the estate, to reduce resource competition and preserve habitat heterogeneity. In Alfina, enhancing habitat suitability for disturbance-sensitive species may similarly require keeping grazing intensity low.
Third, diverse wildlife communities in both areas are associated with landscape heterogeneity. At both sites, rewilding initiatives should aim to retain and enhance such habitat heterogeneity, ensuring that management actions do not excessively favour one functional group at the expense of others.
Fourth, in both sites additional surveys could be performed to assess compositional and topographic data of open habitats to estimate their pastoral value and sustainable stocking rates.

The estimates presented here constitute a robust, pre-rewilding baseline that will enable explicit pre-post comparisons of rewilding outcomes. Temporal changes in density, occupancy, and activity can be used to quantify population-level responses to rewilding, both spatially and temporally. Importantly, changes in activity patterns may provide early indicators of rewilding

impacts, preceding detectable shifts in density or occupancy. A comprehensive monitoring approach is essential to ensure that trophic rewilding initiatives deliver measurable ecological outcomes and can be adaptively managed in response to observed demographic and behavioural changes over time.

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Author contributions

MDM, LS, DM and CD conceived the ideas and designed the methodology. NC, GM, DM and CD collected the data. NC and GM analysed the data. All the authors contributed to the writing, reviewing, editing and finalising of the manuscript.

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Data availability

All population estimates are published with this article, the raw camera trap datasets include sensitive information on wildlife species presence and are available from the corresponding author upon reasonable request.

Competing interests

The authors declare that they have no competing interests.

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