




## Article

# Native Carnivore Diversity and Relative Abundance in Landscapes of the Coast Range in Central Chile: Insights for Conservation Decision-Making

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**Abstract:** Natural habitats in South America have been intensively used and modified, including their conversion to exotic tree plantations, impacting the terrestrial fauna communities. Carnivores play an important role in the functioning of ecosystems as apex predators. Landscape characteristics and resource availability determine the composition and abundance of co-occurring carnivores. We hypothesized that the diversity and relative abundance of native carnivores varied in relation to the different vegetation macrohabitats. We predicted that native forests support a higher diversity and abundance of native carnivores compared to exotic plantations. Between 1 March 2021 and 31 March 2022, we estimated the species' richness, composition, and relative abundance of terrestrial native carnivores in three different landscapes of the Coast Range in central Chile: (a) monoculture plantation of the exotic *Pinus radiata* (MP); (b) Mediterranean coastal thorn forest (TF); and (c) Mediterranean coastal sclerophyllous forest (SF). We monitored an area of 1000 ha in each landscape, installing 10 camera traps (total of 30 camera traps). We used a monitoring transect with unbaited camera traps separated by 1 km, with a total photo-trapping effort of 10,046 camera days. The classification, organization, and analysis of camera trap data were conducted using CameraSweet software. The total number of independent native carnivore species events (photos separated > 60 min) recorded in each landscape were 1564 in SF, 1412 in TF, and 775 in MP. Carnivore richness and composition were not significantly different in all three landscapes. We detected five native carnivore species. Relative abundance by species was significantly higher in SF compared to MP for *Leopardus guigna*, *Conepatus chinga*, and *Galictis cuja* and also significantly higher in TF compared to MP for *L. guigna*. Our results suggest that the native vegetation cover must be preserved to maintain viable and abundant native carnivore populations, crucially relevant in the highly impacted and human-dominated global biodiversity hotspot of Central Chile. This study contributes to informing evidence-based decision-making and conservation strategies at the landscape level to mitigate biodiversity loss.

**Keywords:** landscape use; community ecology; habitat generalist; habitat specialist; land use change; natural habitat conservation



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## 1. Introduction

Natural habitats in South America have been intensively used and modified [1], including their conversion to exotic tree plantations. This phenomenon, in expansion in South America [2,3], changes the natural features of the original ecosystem, impacting the terrestrial fauna communities [2]. The negative effects of this conversion have been

described, such as a lower mammal species richness associated with exotic plantations compared to native vegetation, and also to landscapes with a lower natural forest cover [1,2]. In Chile, this situation is similar to what takes place in the rest of South America [4,5].

The Coast Range is one of the four main macroforms of relief in the geography of Chile; it runs in a north—south direction. It extends for approximately 3000 km, separating the intermediate depression from the coastal plains [6]. Geologically older than the Andes Range, it is a descending formation whose highest summit reaches 3114 m [7]. This mountain range presents different types of ecosystems and a great diversity of fauna, with a high level of endemism and threatened species [8,9]. This area is located in one of the 36 global biodiversity hotspots, in the Central Chile hotspot called the “Chilean Winter Rainfall-Valdivian Forests”, which is Earth’s most biologically rich yet threatened terrestrial regions, therefore with conservation priority [10,11].

Carnivores play an important role in the functioning of natural ecosystems as apex predators [12,13]. They also act as an umbrella species in conservation because they require large areas of suitable habitat to maintain viable populations, therefore protecting these areas benefits many other species and natural communities with smaller habitat requirements (i.e., smaller home ranges) [14–16]. In different parts of the world, sympatric species of carnivores have coexisted over an evolutionary time scale, therefore niche differentiation had already occurred and competition is thus difficult to observe [17]. However, populations of different carnivores interact with each other in a variety of ways and conservation efforts require a greater understanding of the complex relationships between species at the landscape level [17].

Different landscapes can be used selectively by certain species of carnivores, thus producing specialization in the use of the habitat which, therefore, could constitute a limiting resource for the settlement of their populations [18]. Habitat generalists can use various landscapes without limitations for their settlement [18]. Habitat types and their characteristics (e.g., quality and structure) are important factors for the persistence of carnivore populations [19,20]; new methodologies in the modeling of carnivore occupancy provide relevant data towards understanding the multiscale nature of habitat use patterns [21,22]. Some other species-specific factors like movement capacity and individual characteristics (e.g., sex or age) also influence the response of carnivores to habitat heterogeneity [23,24].

Understanding the occurrence of terrestrial carnivores in different landscapes, as well as the interactions and ecological dynamics among them, is essential to proposing strategies for the conservation of these species in the territories [18,25]. When carnivores live in sympatry, some dimensions of their ecological niche can overlap, but coexistence is possible thanks to segregation strategies, with the differential use of space and time being one of the most frequent [26,27]. The landscape characteristics and available habitats model the composition of the carnivores that use them according to the availability of their limited resources. Carnivore composition generates different attributes or ecological parameters for this community ensemble [28], even in intervened human-dominated landscapes or those composed of exotic tree plantations [29,30].

At least seven terrestrial carnivores have been described occupying the central Chile Coast Range: Canidae: *Lycalopex griseus* and *L. culpaeus*; Felidae: *Puma concolor*, *Leopardus guigna*, and *Leopardus colocola*; Mephitidae: *Conepatus chinga*; and Mustelidae: *Galictis cuja* [31]. *L. griseus* and *L. culpaeus* are distributed along the Chilean territory and are diet and habitat generalists [31–34] (mean home range of 1–2.8 and 3.7 km<sup>2</sup>, respectively [33,35]). In recent years, some authors have even proposed the existence of hybridization within the genus *Lycalopex* [36–38]. With a distribution all along the Chilean territory and traditionally linked to the Andes Range, *Puma concolor* has scarce but healthy populations in the Coast Range of the O’Higgins region [39] (mean home range of 550 km<sup>2</sup> [40]). *L. guigna* is distributed latitudinally in Chile between the Coquimbo (29°02′) and Aysén (49°16′) regions, with two subspecies: *L. g. tigrillo* (northern subspecies, assessed in this study) and *L. g. guigna* (southern subspecies) [41–43] (mean home range of 2.7 km<sup>2</sup> [44]). It is a habitat specialist closely associated with the native Mediterranean forests and temperate

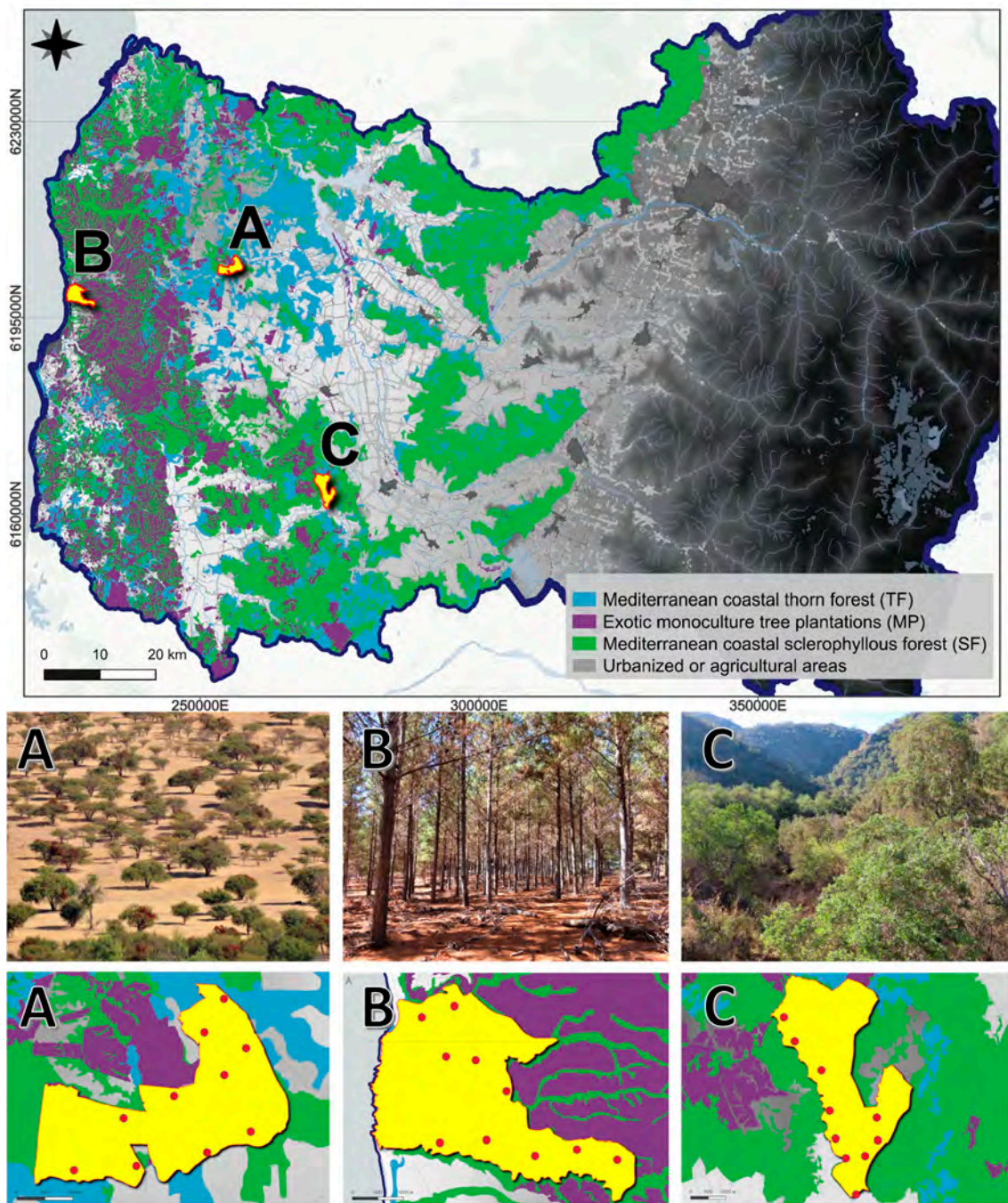
rainforests [44–46] and tolerates human-altered and fragmented landscapes, using small forest fragments and vegetation corridors to move across the human-dominated agricultural matrix [42,47–49] and also using the monoculture pine plantations [50]. The pampas cat, *L. colocola*, is distributed across South America and displays genetic and ecological niche differences along its range [51,52]. It is described along the entire Chilean territory as using diverse habitats [53], but it is scarce and hard to record in central Chile (mean home range of 14.9 km<sup>2</sup> [54]). *C. chinga*, with a crepuscular-nocturnal habit (mean home range of 1.8 km<sup>2</sup> [55]) and *G. cuja*, with a diurnal habit (no home range information available) are scarcely studied in Chile, but with a wide distribution in South America, they occupy a wide diversity of environments being generalists in terms of diet and habitat, but with a suggested niche segregation between them [56,57].

In this study, we assessed the richness, composition, and relative abundance of native carnivores in three different landscape types or vegetation macrohabitats of the central Chile Coast Range: two native forests and one exotic monoculture tree plantation. Our general hypothesis was that the diversity and relative abundance of native carnivores varied in relation to the different vegetation macrohabitats. We predicted that native forests support a higher diversity and abundance of native carnivores compared to exotic plantations. We aimed to obtain inferences on the carnivores' landscape use in native and exotic vegetation covers so we can inform adequate management and conservation plans for sustaining healthy carnivore populations in the highly human-impacted Coast Range of central Chile.

## 2. Methods

There are three main landscapes or vegetation macrohabitats in the Coast Range of O'Higgins region: (a) the Mediterranean coastal sclerophyllous forest (SF) (composed mainly of *Lithrea caustica*, *Quillaja saponaria*, *Cryptocarya alba*, *Peumus boldus*, and *Azara integrifolia*), occupying an area of 257,545 ha; (b) the Mediterranean coastal thorn forest (TF) (composed mainly of *Acacia caven*, *Maytenus boaria*, *Trevoa trinervis*, and *Talguenea quinquinervia*), occupying an area of 130,428 ha; and (c) the exotic monoculture tree plantations (MP) (*Pinus radiata* and *Eucalyptus globulus*), occupying an area of 123,029 ha [58,59]. The first two native vegetation areas have shrunk and are highly fragmented due to their conversion to the introduced exotic monoculture tree plantations, which have gradually increased their area since the 1970s, negatively impacting the native biodiversity [60–62].

We studied these three different landscape types in the following locations: (A) the La Estrella site, composed of native Mediterranean coastal thorn forest (TF); (B) the Alto Colorado site, composed of exotic monoculture tree plantations, *Pinus radiata*, 17 years being the average age at a density of 1600 trees/ha (MP); and (C) the Callihue site, composed of native Mediterranean coastal sclerophyllous forest (SF) (Figure 1). Distance between the study areas ranges from 28 to 56 km; therefore, they are considered as independent. In each of these locations, we selected an area of 1000 ha with a homogeneous and continuous composition of its representative and dominant vegetation type. In each area, considering accessibility (trails), we distributed ten camera traps, separated by 1000 m each, in uniform monitoring transects [63] (Figure 1). At each installation point, we oriented each camera towards trails, passageways, or other areas with attributes indicating a potential fauna passage. No bait or lure was used. We used a Bushnell 24MP Trophy Cam, model 119719CW, with the following settings: mode: camera, image size: HD pixel, capture number: 2 photos, interval: 3 S, sensor level: auto, camera mode: 24 h. The study was conducted during the period between 1 March 2021 and 31 March 2022. Within this period, the cameras were active for 334.87 effective days, with a total photo-trapping effort of 10,046 camera days (3098 in TF, 3446 in MP, and 3502 in SF). We checked the cameras in the field every two months, replacing the batteries. Photos obtained were inspected visually, and only those recording native carnivores were selected for further analysis.



**Figure 1.** Three different sampled landscapes in the Coast Range of the O’Higgins region in central Chile. Upper row: (A) Mediterranean coastal thorn forest (TF), La Estrella site; (B) exotic monoculture tree plantations (MP), Alto Colorado site; (C) Mediterranean coastal sclerophyllous forest (SF), Callihue site. Middle row: Pictures of the three studied landscapes. Bottom row: Red dots show camera trap distribution in the three sites.

We used the CameraSweet programs developed by Sanderson and Harris [64] for classification, organization, and analysis of camera trap data, freely downloaded from the Small Wild Cat Conservation Foundation website [65]. We ran the SpecialReNamer, DataOrganize, and DataAnalyze programs following the methodology described by Vázquez-Ibarra et al. [66]. We determined photos as independent events when separated by >60 min.

Morphological differences between *L. griseus* and *L. culpaeus* are well described [31]. However, the use of camera traps to identify *Lycalopex* species can generate errors in

animals of similar morphology, animals with confusing characteristics, and/or under certain adverse visual environmental conditions [67–69]. To avoid mistakes in the specific identification and classification of these native foxes, since they are nearly indistinguishable via visual analysis of some photographs (mainly at night), and based on their relatively similar ecologies, we grouped the two species functionally as *Lycalopex* spp. canids.

Richness was calculated for each landscape type by counting the number of different carnivore species detected. Composition, the identity of the detected carnivore species, was described. Species abundance was expressed as relative abundance index (RAI). RAIs were calculated for all the detected species collectively and separately, for each landscape, as the sum of independent events recorded and divided by the sum of sampling efforts. The relationships of all-species RAI and species-specific RAIs with landscape was explored through bar graphs.

Significant differences in the total number of carnivores and in the number of specific carnivore species across the three landscapes were evaluated using regression models for count data, with the number of recorded events as the response variable, the landscape as the unique predictor, and the sampling effort as the exposure. Models were built on site-level (i.e., camera) data points ( $n = 30$ ), according to the following general equation:

$$\ln E(Y) = \ln(n) + \beta_0 + \beta_1 X$$

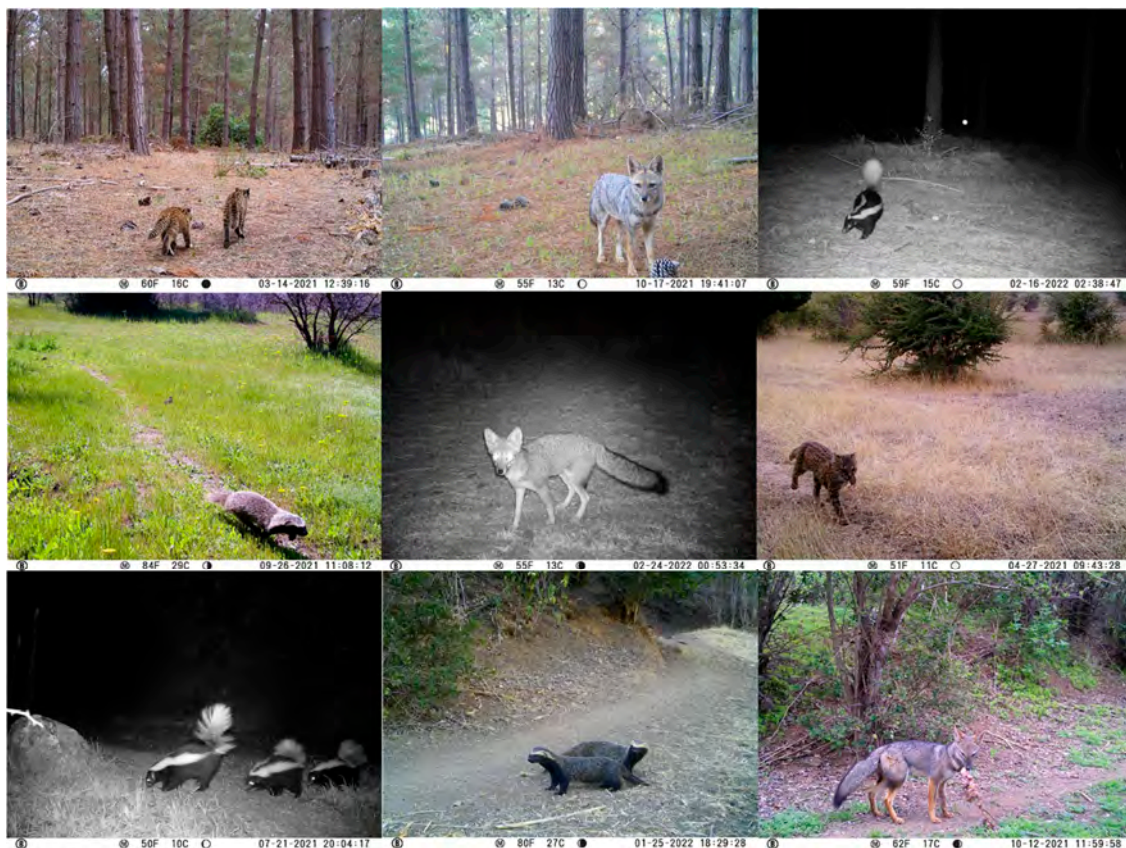
where  $\ln E(Y)$  is the log of the expected event count,  $\ln(n)$  is the log of the sampling effort,  $\beta_0$  is the intercept,  $X$  is the vector of the predictor values (i.e., landscape), and  $\beta_1$  is its corresponding vector of coefficients. In each evaluation, we first adjusted a Poisson regression model whose overall fit was assessed through the deviance  $\chi^2$  goodness-of-fit test, while the presence of overdispersion was computed as the deviance  $\chi^2$  divided by its degrees of freedom. If the goodness-of-fit test was not significant (suggesting a lack-of-fit) or the overdispersion was greater than 1.25, indicating that the Poisson model is not valid [70], we fit a negative binomial (NB) model. The hypothesis that the NB dispersion parameter  $\alpha > 0$  (i.e., the NB model is more appropriate than the Poisson model) was tested using a likelihood ratio test (LRT). All the Poisson and NB models were fitted using a full maximum likelihood (ML) estimation. The Poisson model and NB model diagnostics were carried out by plotting deviance residuals vs. predicted counts and by inspecting for any obvious pattern. If a high proportion of zeros (>30%) or lack-of-fit was present, or the unaccounted overdispersion was still above 1.25 in the NB model, we adjusted for a zero-inflated negative binomial (ZINB) and zero-inflated Poisson (ZIP) models using the landscape predictor in both the binary and the count parts of the models. We used the Akaike information criteria (AIC) to confirm the selection of the best model among the NB, ZINB, and ZIP models. We produced five final models: one for all carnivores (including all the detected species together) and one for each of the species. Pairwise comparisons between the landscapes were performed and adjusted using the Bonferroni method. Assuming a global significance level of  $p = 0.05$  and three possible pairwise comparisons, the penalized significance level was  $p = 0.017$ . We used the final models to predict the number of events across the three landscapes, with the sampling effort set at their mean value. All statistical analyses were carried out in Stata v15 (StataCorp, College Station, TX, USA).

### 3. Results

We recorded a total of 1564 independent native carnivore events for all species in SF, 1412 in TF, and 775 in MP. Five native carnivore species were detected, co-occurring at the three study sites: *L. guigna*, *C. chinga*, *G. cuja*, *L. griseus*, and *L. culpaeus* (Table 1, Figure 2).

**Table 1.** Number of independent events recorded (n), sampling effort (days × camera), and mean relative abundance index (RAI = n/days) of native carnivore species in three different landscapes from the Coast Range of central Chile.

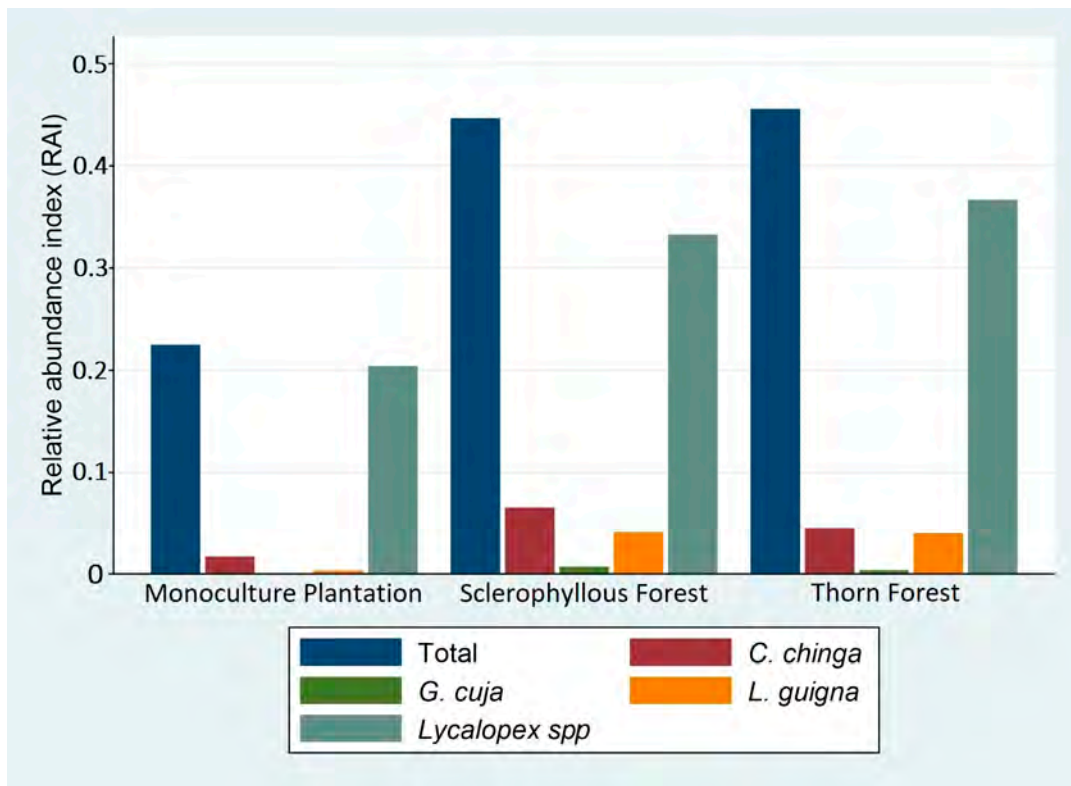
| Location      | Landscape                        | Total Sampling Effort (Days × Camera) | N° of Cameras | Total Number of Independent Events (RAI = n/days) |                |                  |                       |                  |
|---------------|----------------------------------|---------------------------------------|---------------|---|----------------|------------------|-----------------------|------------------|
|               |                                  |                                       |               | <i>C. chinga</i>                                  | <i>G. cuja</i> | <i>L. guigna</i> | <i>Lycalopex</i> spp. | All Species      |
| Alto Colorado | Monoculture tree plantation (MP) | 3446                                  | 10            | 59<br>(0.0171)                                    | 2<br>(0.006)   | 11<br>(0.0032)   | 703<br>(0.2040)       | 775<br>(0.2249)  |
| Callihue      | Sclerophyllous forest (SF)       | 3502                                  | 10            | 230<br>(0.0657)                                   | 25<br>(0.0071) | 144<br>(0.0411)  | 1165<br>(0.3327)      | 1564<br>(0.4466) |
| La Estrella   | Thorn forest (TF)                | 3098                                  | 10            | 141<br>(0.0455)                                   | 11<br>(0.0036) | 124<br>(0.0400)  | 1136<br>(0.3667)      | 1412<br>(0.4558) |



**Figure 2.** Camera trap pictures of native carnivores recorded in three different landscapes from the Coast Range of the O’Higgins region in central Chile. Upper row: Monoculture plantation of exotic *Pinus radiata* (MP) (from left to right: *Leopardus guigna*, *Lycalopex* sp., and *Conepatus chinga*); middle row: Mediterranean coastal thorn forest (TF) (*Galictis cuja*, *Lycalopex* sp., and *L. guigna*); bottom row: Mediterranean coastal sclerophyllous forest (SF) (*C. chinga*, *G. cuja*, and *Lycalopex* sp.).

The *Lycalopex* species group had the highest total number of independent events and the highest mean relative abundance index (RAI = n/days) in the three landscapes (Table 1, Figure 3). The *Lycalopex* were mostly observed in the thorn forest (RAI = 0.3667), followed by the sclerophyllous forest (RAI = 0.3327), and the monoculture plantations (RAI = 0.2040). The number of *Lycalopex* events were best fit with a NB model. The model showed no significant differences in the total number of events recorded among landscapes (Table 2, Figure 4). Of the photographs of foxes that could be confidently identified ( $n = 183$ ), 74%

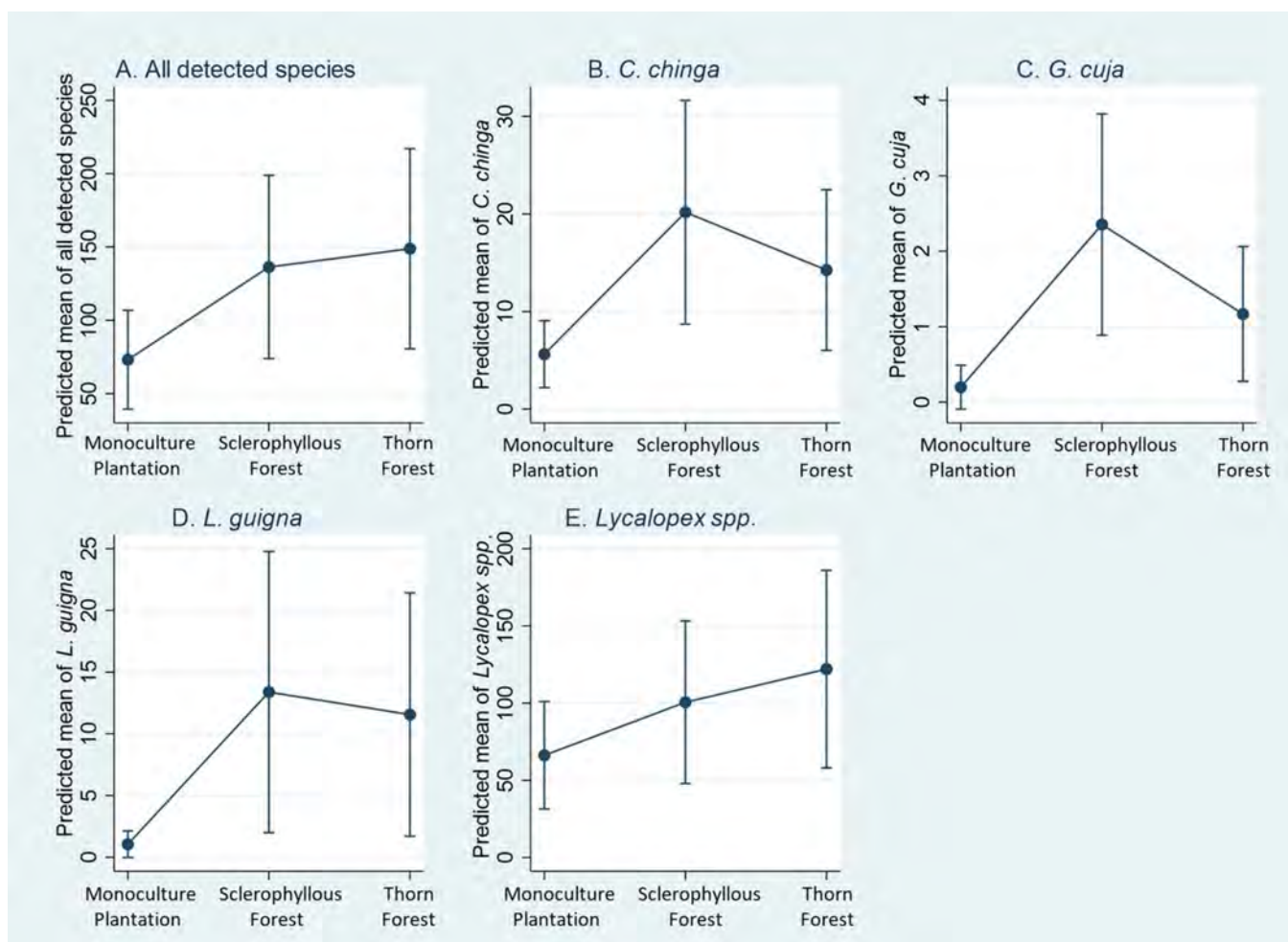
( $n = 136$ ) could be identified as *L. griseus* and the remaining 26% ( $n = 47$ ) were identified as *L. culpaeus*.



**Figure 3.** Relative abundance indices of native carnivore species in three different landscapes from the Coast Range of the O’Higgins region in central Chile. *G. cuja* = *Galictis cuja*; *C. chinga* = *Conepatus chinga*; and *L. guigna* = *Leopardus guigna*.

**Table 2.** Final negative binomial models for the event counts for all species and for each species detected with location as the unique predictor. Coefficients, standard errors (in parentheses), and *p*-values (in *italics*) for locations and for the dispersion parameter alpha are shown ( $n = 30$ ). The log of the sampling efforts is included as the offset. Significant *p*-values are shown in bold.

| Variable Name                             | All Detected Species                  | <i>C. chinga</i>                      | <i>G. cuja</i>                        | <i>L. guigna</i>                      | <i>Lycalopex spp.</i>                 |
|---|---------------------------------------|---------------------------------------|---------------------------------------|---------------------------------------|---------------------------------------|
| Intercept                                 | −1.522<br>(0.235)<br><b>&lt;0.001</b> | −4.085<br>(0.311)<br><b>&lt;0.001</b> | −7.431<br>(0.749)<br><b>&lt;0.001</b> | −5.763<br>(0.522)<br><b>&lt;0.001</b> | −1.621<br>(0.268)<br><b>&lt;0.001</b> |
| Location                                  | 0.066                                 | <b>0.009</b>                          | <b>0.008</b>                          | <b>&lt;0.001</b>                      | 0.256                                 |
| (Monoculture Plantations (MP) (reference) | --                                    | --                                    | --                                    | --                                    | --                                    |
| Sclerophyllous Forest (SF)                | 0.623<br>(0.332)<br>0.060             | 1.276<br>(0.425)<br><b>0.003</b>      | 2.473<br>(0.814)<br><b>0.002</b>      | 2.544<br>(0.679)<br><b>&lt;0.001</b>  | 0.418<br>(0.378)<br>0.269             |
| Thorn Forest (TF)                         | 0.711<br>(0.332)<br><b>0.032</b>      | 0.929<br>(0.428)<br><b>0.030</b>      | 1.772<br>(0.845)<br><b>0.036</b>      | 2.397<br>(0.680)<br><b>&lt;0.001</b>  | 0.612<br>(0.379)<br>0.106             |
| Alpha                                     | 0.540<br>(0.133)<br><b>&lt;0.001</b>  | 0.796<br>(0.209)<br><b>&lt;0.001</b>  | 0.601<br>(0.423)<br><b>&lt;0.001</b>  | 1.813<br>(0.558)<br><b>&lt;0.001</b>  | 0.705<br>(0.173)<br><b>&lt;0.001</b>  |

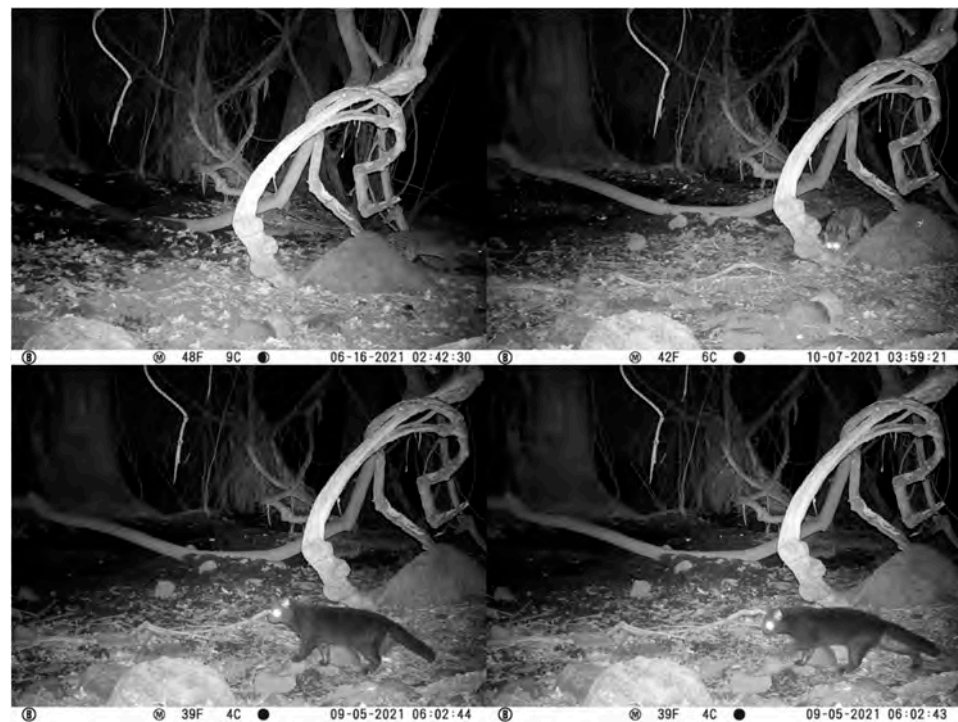


**Figure 4.** Negative binomial model predicted means for native carnivore species in three different landscapes from the Coast Range of the O’Higgins region in central Chile. (A) All carnivore species, (B–E) each carnivore species separately, and their 95% confidence intervals by landscape. Sampling efforts were set at their mean values. *Conepatus chinga*, *Galictis cuja*, *Leopardus guigna*: significantly higher number of records in sclerophyllous forest (SF) followed by thorn forest (TF), compared to significantly lower number of detections in monoculture plantations (MP). *Lycalopex sp.*: non-significant differences. Significant *p*-values are shown in Table 2.

The other carnivores recorded, *L. guigna*, *C. chinga*, and *G. cuja*, were mainly observed in the sclerophyllous forest (RAI = 0.0411, 0.0657, and 0.0071, respectively), followed by the thorn forest (RAI = 0.0400, 0.0455, and 0.0036, respectively), and the monoculture plantations (RAI = 0.0032, 0.0171, and 0.0006, respectively) (Table 1, Figure 3). The number of *L. guigna*, *C. chinga*, and *G. cuja* recorded events were best fit with a NB model. In the case of *C. chinga* and *G. cuja* the highest numbers were observed in SF, exhibiting significant differences only when compared to MP ( $p = 0.008$ , in both species), but not when compared to TF. *L. guigna* was detected more frequently in SF and TF, presenting significant differences when compared to MP ( $p = 0.001$ , in both landscapes), but not when compared to each other (Table 2, Figure 4). None of the models presented any obvious pattern when the deviance residuals versus predicted counts were plotted, suggesting that apparently there are no substantial unmeasured factors that influence the event counts.

We detected a melanistic individual of *Leopardus guigna tigrillo* at one site in SF (Figure 5).





**Figure 5.** First known record of a melanistic *Leopardus guigna tigrillo* (guigna northern subspecies). Upper row: Guigna, spotted coat pattern; bottom row: Guigna, melanistic coat pattern. Pictures were recorded at the same location site in SF, shown for comparison between the two coat patterns.

#### 4. Discussion

Both species of the genus *Lycalopex* were recorded with a higher relative abundance than the other carnivores in the three studied landscapes. This result shows that this genus is the most abundant of the native carnivores in the Coast Range of central Chile. The *Lycalopex* species did not show any significant differences among the study areas, suggesting that they are generalists in terms of habitat use and diet [34,71–73]. From the total photographs of foxes that could be confidently identified, *L. griseus* was recorded as the most abundant species of native carnivores across our study sites in the Coast Range of central Chile.

The other carnivores, *L. guigna*, *C. chinga*, and *G. cuja*, were more frequently observed in the sclerophyllous forest, followed by the thorn forest, and compared to a significantly lower number of records in the monoculture plantation, suggesting a preference of these carnivores for both native landscapes. Specifically for *G. cuja*, this pattern contrasts with what was found by Zúñiga et al. [71] in southern Chile, who described that this species has a marked preference for fragmented monoculture tree plantations compared to native forests and grasslands. In central Chile, *G. cuja* is more commonly found in the central valley (including pastures or cropland for agricultural and livestock use) than in the Coast Range [74], and according to previous studies, it is considered a habitat generalist [75].

Our study is coincident with previous records showing the ability of *L. guigna* to tolerate different landscapes [45,47,50,76], including exotic monocultures and fruit tree plantations (this study). Some of these habitats may be occupied only as hunting or transit sites between areas of dense native vegetation, where permanent refuges and reproductive sites are located. As with most felids, vegetation cover has been described as an important ecological requirement for *L. guigna*, used for stalking prey and reproduction [77]. However, based on our data, it seems that the presence of dense understory is not always an obligatory ecological requirement for *L. guigna* movements across these landscapes, as shown by several photographs we obtained (i.e., Figure 2, upper left and middle right), where the species appears to be moving in the absence of dense understory both across exotic monoculture plantations and native thorn forests. Regarding the melanistic individual of

*Leopardus guigna tigrillo* detected at one site in SF, to our knowledge, this is the first record of a melanistic coat pattern in guignas in the northern subspecies' range [41–43,78].

Two potentially distributed species were not detected in our study: *P. concolor* and *L. colocola*. For *L. colocola*, it may be due to its low abundance and scarce populations endemic to central Chile; its conservation status has been suggested to be upgraded to categories of higher threat levels according to the latest taxonomic assessments [52,79–81]. *P. concolor* is a species whose presence has been previously recorded in these landscapes of the Coast Range of central Chile [39]. However, the lack of detection in this study is coincidental with its cryptic movements and low detection rates, even with a high sampling effort. Recently, Garcia et al. [16] studied the presence of carnivores for 4 years (2013–2016), installing 53 camera traps (baited with lynx urine) in 12 remnant patches of sclerophyllous forest and shrublands in vineyard landscapes of the Mediterranean region of central Chile, but did not record *P. concolor*.

The presence of these five different carnivores co-occupying the same landscapes in the Coast Range allows us to infer that the heterogeneity of the landscapes and sufficient resources for their subsistence probably facilitates their co-existence [82–84]. Habitat preference and the dispersal of carnivores from their source reproductive areas to other areas with less availability of resources may follow the source–sink model of population dynamics [85], which predicts that density drives the emigration of subordinate animals to habitats offering lower competition for resources [86]. Our results show that native carnivores recorded here are either habitat generalists [71] or can adapt to inhabiting human-dominated landscapes using the remnant available vegetation cover in central Chile. Even fruit tree plantations or monoculture tree plantations of exotic species (*Pinus radiata*) with or without dense understory can provide cover and/or corridors for their lurking predation habits, and abundant populations of lagomorphs, rodents, and invertebrates in these sites may constitute their local main diet. According to Coon et al. [87], prey vulnerability may be more important than prey availability, following what Hopcraft et al. [88] called the 'ambush–habitat hypothesis', which proposes that predators choose habitats not based on prey density or encounter rates but on other factors such as prey vulnerability and the minimization of bodily risk. However, to the best of our knowledge, there are no studies on the abundance or density of potential prey for these predators in the area.

Monoculture tree plantation landscapes are temporal habitats and clearcutting can have negative effects for the animals occupying these landscapes [89,90]. Previous studies have recorded that species with more specific habitat requirements (habitat specialists), such as *L. guigna*, are unlikely to move through clear cuts and young plantations, but adult pine plantations with or without understory enables their movements across the landscape [50,90]. It has been previously described that pine plantations are suitable for habitat generalists such as *C. chinga* and *L. culpaeus* [90]. Other studies in South America have explored mammal richness in relation to the age of exotic tree plantations. A study by de Abreu et al. [2] proposed that Eucalyptus plantations at early ages are more used by mammals due to their shrubby aspect, a positive visual effect for resources and shelter, understory, and lower anthropogenic disturbance. Cravino et al. [3] found that in Eucalyptus plantations at intermediate ages (2–4 years), when the vegetation complexity is similar to the structure of a native forest, mammal species richness was at its maximum, but still lower compared to natural forest habitats.

Some studies suggest that the biodiversity is greater in the Coast Range than in the Andes at a similar latitude [91,92]. However, and in contrast, the wild areas protected by the Chilean state (SNASPE) are notably larger in the Andes Range than in the Coast Range [9]. At least, the O'Higgins region in central Chile (34–35° S) has 38,582 ha of SNASPE in the Andes, but only 3715 ha in the Coast Range [93]. Therefore, areas with high biodiversity in Chile, such as the Coast Range, currently do not have enough legal protection. Also, carnivores have relatively wide home ranges and are territorial (adult males maintain exclusive intraspecific territories; female home ranges occur within male home ranges); therefore, our records probably correspond mainly to resident individuals

(with the exception of dispersing subadults). In this context, it is urgently necessary to establish landscape-level conservation strategies for these communities of native carnivores inhabiting human-dominated landscapes, to help maintain a healthy ecosystem in the Coast Range of central Chile. In support of our predictions, our results reveal that the native vegetation cover harbors a significantly higher relative abundance of native carnivores compared to exotic pine plantations. Therefore, we recommend that both the sclerophyll and thorn native forest remnants should be preserved to maintain viable and abundant native carnivore populations, as has been recorded in other studies in Chile [15,76]. In other areas of South America, Ferreira et al. [1] have highlighted the imperative of maintaining forest remnants to retain the forest-dwelling mammals in human-dominated landscapes.

Conservation strategies in these landscapes should also protect vegetation corridors, which are crucial for dispersal among subpopulations to secure the supply of immigrants and thus contribute to metapopulation persistence [94], especially in the case of subpopulations affected by fragmentation or offtake by humans [90,95]. These strategies should also include the implementation of human–carnivore coexistence formulas, allowing the sustainable balance between habitability and human development and the conservation of healthy native carnivore populations, being one of the most difficult challenges in the currently increasing global change scenario.

Different conservation contexts and strategies at the local scale can provide insights for comparison and extrapolation to solve conservation problems in other parts of the world. This study contributes to the discussion of the challenges of carnivore conservation in human-dominated landscapes, connected to global conservation issues. Given that information about carnivore communities in these ecosystems is scarce, our results contribute towards building up valuable baseline information for evidence-based decision-making and conservation planning to mitigate biodiversity loss. We highlight the importance of the central Chilean Coastal Range native forests as an important source and reserve of healthy carnivore populations, which is crucially relevant in the highly impacted and human-dominated global biodiversity hotspot of Central Chile.

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