Global Ecology and Conservation 23 (2020) e01108

Contents lists available at ScienceDirect

Global Ecology and Conservation

journal homepage: http://www.elsevier.com/locate/gecco

Original Research Article

Defining corridors for movement of multiple species in a forest-plantation landscape

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ARTICLE INFO

Article history:
Received 14 April 2020
Received in revised form 8 May 2020
Accepted 8 May 2020

Keywords:
Central Chile
Forestry landscapes
Spatial priorization
Habitat quality
Connectivity

ABSTRACT

Forestry plantations of *Pinus radiata*, managed through clearcutting method, would offer resistance to the movement of native fauna among remnants of native forest. In this study, we evaluate the possibilities of establishing connectivity corridors for native fauna in a forest landscape dominated by *P. radiata*, focusing on seven species with some level of specialisation in the use of different land covers and that exhibit different movement strategies. We sampled presences and abundances using several types of sampling to determine the abundance and crossing edge probabilities for the ground-dwelling beetle *Ceroglossus chilensis*; the mesocarnivores *Conepatus chinga, Lycalopex culpaeus* and *Leopardus guigna*; the terrestrial frogs *Eupsophus septentrionalis* and *Pleuroderma thaul*, and the understory bird *Pteroptochos castaneus*. We use species distribution modelling and circuit theory to calculate connectivity for each species. Finally, a spatial priorization approach was applied to obtain two scenarios of least-cost corridors. The results show that species with more specific requirements such as *L. guigna, P. castaneus, P. thaul* and *E. septentrionalis*, are unlikely to move through clear cuttings and young plantations, but their movement is possible through adult pine plantations with understory. For species that are habitat generalists such as *C. chinga and L. culpaeus* plantations were suitable for movement. The management of forest plantations should use multi-taxa assessments in the design of biological corridors that allow the connection of forest fragments.

1. Introduction

Habitat fragmentation is one of the greatest threats to biodiversity worldwide (Fahrig, 2003; Ewers and Didham, 2006; Haddad et al., 2015) that often leads remnant habitat to be surrounded by a hostile matrix acting as a barrier to dispersal migration and gene flow (Eycott et al., 2012). In a changing world, maintenance and restoration of landscape connectivity

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https://doi.org/10.1016/j.gecco.2020.e01108

have become conservation priorities to mitigate impacts of intensive forestry, agricultural practices and urban land conversion (Crooks and Sanjayan, 2006; de Chazal and Rounsevell, 2009; Hawn et al., 2018). However, for several species, the reduced information available about their dispersion or habitat requirements to move prevents implementation of sustainable landscape planning (Bowne and Bowers, 2004). In this sense, empirical data are necessary to evaluate if species use anthropogenic lands as corridors for movement between habitat fragments. For example, carnivores' occurrence data from camera trapping has improved our understanding of how frequent riparian habitat and patches of complex vegetation are used for movement towards protected areas (Hilty and Merenlender, 2004; Moreira-Arce et al., 2016). Indeed, metrics of landscape connectivity can be derived from spatial models based on wildlife occurrence patterns with these modelling approaches providing a quantitative basis to identify potential corridors in fragmented landscapes (Pe'er et al., 2011).

Spatially explicit models, such as least-cost path mapping (Sawyer et al., 2011) and toolkits for GIS (CorridorDesign, 2017), have been gaining attention as approaches to establish movement corridors at the regional and landscape levels. One of the most novel applications using landscape-scale empirical data of wildlife is the electric circuit theory (McRae et al., 2008). In this approach, voltage and resistance are, respectively, the probability of occurrence in a single patch and the probability that one species crosses the limit between two different patches (McRae et al., 2008). Spatially-explicit models based on electric circuit theory are being increasingly applied in the design of regional-scale corridors for several taxa and different ecosystems (Braaker et al., 2014; Grafius et al., 2017; Merrick and Koprowski, 2017; Naidoo et al., 2018; Dai et al., 2019). However, these models have been poorly used in temperate forest landscapes of the Southern Hemisphere, whose rich biodiversity is particularly vulnerable to increasing pressures from agriculture and more recently, from forestry industry (Schaffer-Smith et al., 2016; Thompson and Velilla, 2017). Electric-circuit-based modelling offers appropriate tools to quantify the functional connectivity of industrial forestry landscapes. In these landscapes, multiple species with different movement abilities and specific habitat preferences coexist in a mosaic of native forest and forest stands to differ in age and vegetation structure/composition.

Industrial forestry is responsible for reducing species richness worldwide by 13–75% while generating several negative ecological impacts, especially in the movement and dispersion of several taxa (Crook et al., 2011; Dixo et al., 2009; Fahrig, 2003; Haddad et al., 2015; Vergara and Simonetti, 2006). Public, governmental, international and stakeholder pressure is leading to forest companies to adopt sustainable harvesting practices intended to prevent managed lands to become unsuitable habitats and barriers for wildlife movement, as stated in the Aichi' targets (Paquette and Messier, 2009; UN, 2010). Sustainable forestry practices are required to conserve understory birds and ground-dwelling beetles, whose movements are limited by Monterrey Pine plantations with sparse understory vegetation. Standard forestry practices in Chile follow the extensive application of the clearcutting system (Vergara and Simonetti, 2006; Cerda et al., 2015; Ramírez-Collío et al., 2017; Russek et al., 2017). Monterrey Pine plantations in Chile, which cover ca. 2 million ha (INFOR, 2010), are the main driver of the Maulino native forest replacement (Echeverría et al., 2006; Estades et al., 2012; Miranda et al., 2017). The Maulino forest is a unique ecosystem located in the coastal range of south central Chile, from 35°55 to 36°20 S, conformed by many endemic animal and trees species and considered an endangered ecosystem because it has been heavily fragmented and replaced historically by agricultural and forestry industry (San Martin and Donoso, 1996; Smith-Ramírez et al., 2004). However, such a native forest replacement by pine plantations not necessarily has resulted in a sharp decline of biodiversity. Indeed, several animal species occupy forest fragments and use pine plantations as an alternative habitat to feed or move between forest fragments, including the threatened Los Queules toad (*Eupsophus septeptrionalis*), kodkod cat (*Leopardus guigna*), growndwelling beetle (*Ceroglossus chilensis*) and tapaculos (*Pteroptochos castaneus*) (Acosta-Jamett and Simonetti, 2004; Cerda et al., 2015; Puente-Torres et al., 2017, 2017; Ramírez-Collío et al., 2017; Russek et al., 2017; Vergara and Simonetti, 2006).

The conservation of wildlife species in pine plantation stands usually focus on maintaining a dense understory cover with the aim to improves habitat quality. Thus, providing breeding sites, anti-predatory protection and microhabitats rich in feeding resources (Acosta-Jamett and Simonetti, 2004; Vergara and Simonetti, 2006; Henriquez et al., 2009; Simonetti et al., 2013; Ramírez-Collío et al., 2017; Russek et al., 2017). However, the impacts of landscape-scale planning of forestry practices and the design of wildlife corridors remain poorly studied in landscapes dominated by forest plantations.

In this study, we used species distribution modelling, electric-circuit-based modelling and spatial priorization to quantify the functional connectivity of different animal species in a Chilean Mediterranean forest (Maulino forest) landscape fragmented by plantations of the exotic Monterrey Pine (*Pinus radiata*). Achieving the Aichi Biodiversity Target of ensuring forestry landscapes are functionally connected for wildlife requires evaluating the connectivity for multiple species differing in their life history, habitat preferences and movement abilities (Brodie et al., 2015; Ayram et al., 2018; Ersoy et al., 2018). Such a multi-taxa approach may provide insights into the sustainable landscape-scale management of the forest matrix to maintain the viability of multiple metapopulations. To address these issues, we developed a methodological framework intended to identify forest management options based on multi-taxa connectivity patterns.

2. Material and methods

2.1. Study site

Our study was conducted in the coastal range of central Chile (72° 40′W, 35° 60′S), around Los Queules National Reserve (35° 59′ S, 72° 41′ W). The landscape comprises fragments of deciduous Maulino forest (<100 ha) surrounded by plantations of Monterrey pine harvested under the clearcutting system, with rotations of 15–20yr (Vergara and Simonetti, 2004, Fig. 1).

Maulino forest is one of the most threatened ecosystems of Chile (Pliscoff, 2015; Luebert and Pliscoff, 2017), being part of the Chilean Winter Rainfall-Valdivian Forests, one of the 34 world biodiversity Hotspots (Myers et al., 2000). The Maulino Forest has undergone a reduction in the area of 67% between 1975 and 2000, equivalent to an annual forest loss rate of 4.5% per year, mainly caused by the rapid expansion of forestry (Echeverría et al., 2006; Miranda et al., 2017). Maulino forest is composed by native trees such as *Nothofagus glauca*, *Cryptocarya alba*, *Lithraea caustica*, *Aetoxicum punctatum*, *Peumus boldus* and *Luma apiculata*, while understory is composed by native shrubs such as *Aristotelia chilensis* and *Chusquea* spp. (Luebert and Pliscoff, 2017).

2.2. Species background

Functional connectivity was assessed for seven taxonomically distant animal species of the Maulino forest, differing in their body size, life history traits and preferences for habitat resources and movement capabilities (Simonetti et al., 2018): the ground-dwelling beetle *Ceroglossus chilensis* (2.2–2.9 cm length); two endemic frogs: *Eupsophus septeptrionalis* (3.4–4.7 cm) and *Pleuroderma thaul* (2.6–5.5 cm), the understory bird *Pteroptochos castaneus* (24 cm), and three native mesocarnivores: *Conepatus chinga* (50–60 cm), *Lycalopex culpaeus* (140 cm) and *Leopardus guigna* (54–77 cm). Some of those species (i.e. *E. septeptrionalis* and *L. guigna*) are classified as threatened by IUCN or local Ministry of Environment (www.mma.gob.cl/clasificacionespecies) or protected by national Hunting Law (i.e. *C. chilensis*). Those species also differed widely in their movement capabilities, with some of them exhibiting relatively short movement distances, like *C. chilensis* (up to 100 m per day; Cerda et al., 2015), others with intermediate movement capabilities such as *L. guigna* (up to 1.8 km per day; Freer, 2004), and others with greater ability to move like *L. culpeus* (up to 8 km per day; Salvatori et al., 1999). In similar tapaculos understory birds, its can move 674 m in 28 days (Castellón and Sieving, 2006). Frogs studied not present information available of distance movement.

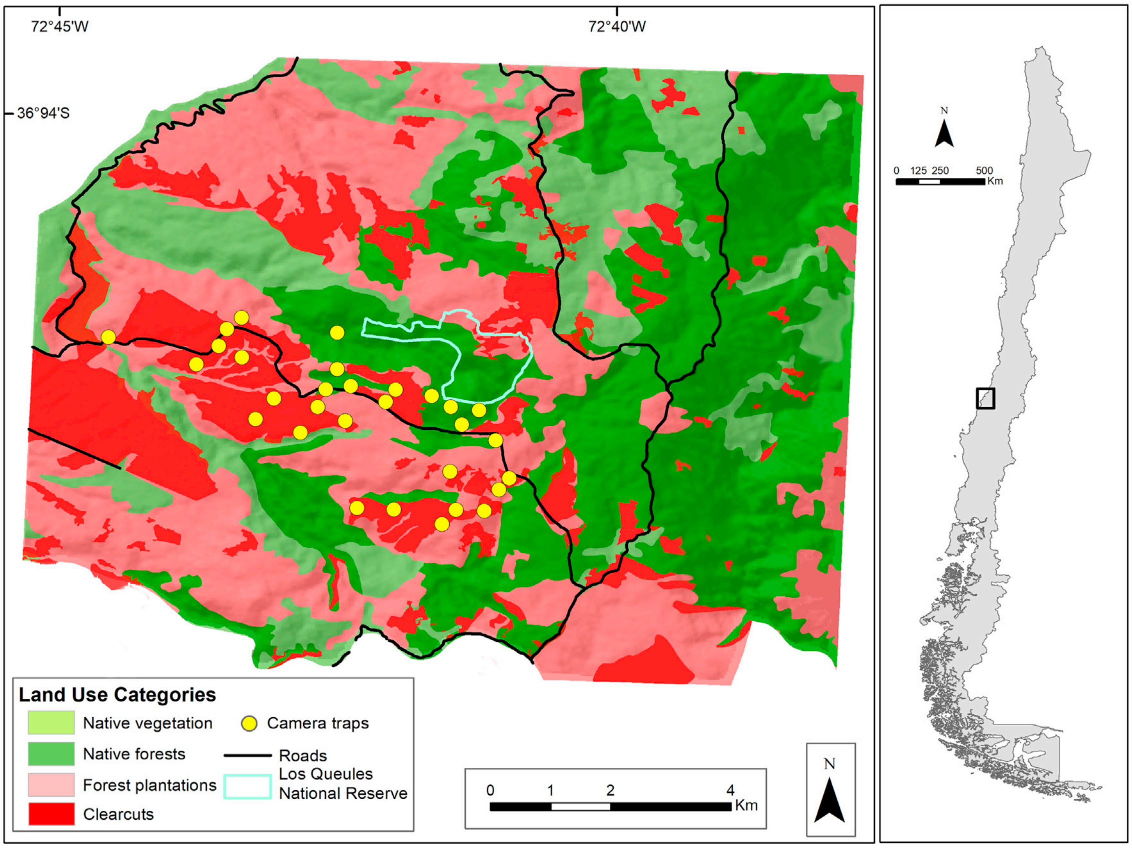

Fig. 1. Study forestry landscape located at the coastal range of central Chile.

2.3. Methodological framework

We developed a step-by-step procedure for identifying potential corridors connecting local populations of species known to be sensitive to forestry land-cover type (Fig. 2). Using occurrence and abundance data of the seven species selected and environmental variables, we obtained first species distribution models. Then, the probability of presence was used to calculate resistance surfaces and defines connectivity models. Finally, species-specific corridors will be derived from a spatial priorization based on connectivity abilities.

2.3.1. Occurrence and abundance of species

Abundance estimates for each species were derived from different sampling methodologies. Abundance and edge cross probabilities for all taxa were obtained for each land cover (native forest (n = 6), pine plantations (n = 6) and clear-cut stands (n = 18)) between patches of the same land cover type (see Table 1). For *C. chilensis* (n = 201), we used the accumulative abundance of individuals captured using nine Barber pitfall traps per plot (n = 6 plots per land cover type; Russek et al., 2017). Traps operated four consecutive days per month between October 2014 and July 2015 (960 h in total) and consisted of plastic jars containing water and detergent as a killing-preserving solution, buried at ground level, and located 10 m apart within a 30 × 30 m grid of 3 × 3 traps (Russek et al., 2017). In the case of the frogs, abundance (*P. thaul* n = 64 individuals and *E. septeptrionalis* n = 58 individual) was obtained monthly with an active count of individuals in five plots per land cover type along 80 m transects. Sampling was performed for four months consecutively (n = 100 samples), and the total sampling effort was of 26.3 h (Puente-Torres et al., 2017; 2017). Total effort per land cover was for native forest 786 min; for pine plantations was 372 min and clear-cut stands 421 min. We used abundance estimates of *P. castaneus* obtained from an N-mixture model presented in Ramírez-Collío et al. (2017) which accounting for imperfect detection. *Pteroptochos castaneus* individuals (n = 96) were recorded from 30 sample points each with a total of 50 m fixed-radius point counts using broadcast survey (Ramírez-Collio et al., 2017). We used four broadcast survey per sample point both breeding and post-breeding period. These consisted of playing vocalisations and territorial songs using a portable speaker for two 1-min periods, each one separated by 3 min to detect birds (1 + 3+1 + 3 = 8 min; Ramírez-Collio et al., 2017). *Pteroptochos castaneus* active surveys began at dawn (05:30 h) and continued until 12:00 h during the breeding period (Sep–Dec 2014, the austral spring) and post-breeding period (Jan–March 2015, the austral summer; see details in Ramírez-Collio et al., 2017). The abundance of carnivores (*L. guigna*: n = 52 individuals; *L. culpaeus*: n = 110 individuals and *C. chinga*: n = 17 individuals) was sampled from camera traps (n = 5 cameras for native forest and pine plantations and 10 for clear-cut stands), which operated in parallel and monthly in different land cover. For mesocarnivores abundance, we used per species the absolute number of visits per sample plot in each land cover (Javier A. Simonetti, unpublished data). Total effort per land cover was for the native forest of 12,643 h; for pine plantations was 10,481 h and for clear-cut stands was 23,874 h. We calculated the probability of edge cross of each species as based on their abundance in each land cover type (the number of species records in each land cover divided by the same total land cover area).

2.3.2. Species distribution models

Maxent 3.3.4 (Phillips et al., 2006, 2017) were used to estimate the probability of presence for the seven-species analysed. We calculate habitat quality for each species calibrating and projecting distribution models inside the study area (Fig. 1), knowing that the study area does not have a large spatial extension (102.5 km2). We chose the maxent method despite other common similar approaches (e.g. GLM, Random Forest) due to the ability to generate fit models with high accuracy based only on known presences (Elith et al., 2010). Also, we decide to use environmental variables at high resolution (25 m) to project the habitat quality models where the total number of pixels in the study area at this resolution is similar to regional scale works at 1 km2 with maxent (111,815 pixels). Species occurrence localities for each species were selected from fragments belonging to the land covers with higher values of abundance (Table 1) to reflect current habitat preferences of species. Once localities areas were selected, fifty occurrences for every seven species were selected across fragments of different land cover with high species abundance. Five environmental variables were used as variables for modelling in the Maxent environment. They were elevation, slope, aspect, topographic position index (calculated using an SRTM dem of 25 m; Farr et al., 2007) and land-cover type use as a categorical variable (using updated data from the National vegetation cadastral of vegetation in CONAF, 1999). Land-cover type map was updated using the last scene available for the area of Google Earth. Clear-cuts areas were identified

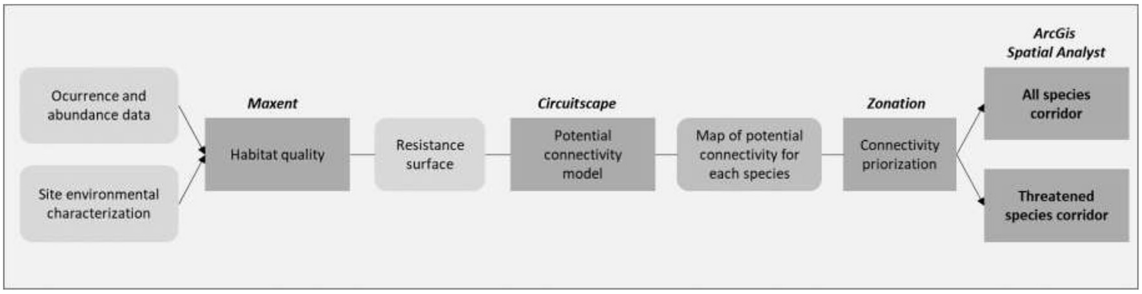

Fig. 2. Methodological steps used in this study to estimate movement corridors for wildlife.

Table 1
Abundance and edge cross probability in native forest, pine plantations and clearcuts, sampling method and references for each species.

Species	Native forest		Pine plantations		Clear cuts		Sample method	References
	Abundance	Probability	Abundance	Probability	Abundance	Probability		
Leopardus guigna	5.33 ± 2.1	0.08 ± 0.02	2.66 ± 0.9	0.09 ± 0.03	0.22 ± 0.1	0.06 ± 0.03	Camera trap	J. Simonetti unpublishing data
Lycalopex culpaeus	4.50 ± 2.4	0.09 ± 0.04	4.66 ± 1.5	0.14 ± 0.03	3.05 ± 0.6	0.47 ± 0.07	Camera trap	J. Simonetti unpublishing data
Conepatus chinga	0.66 ± 0.3	0.01 ± 0.01	0.16 ± 0.1	0.01 ± 0.01	0.66 ± 0.2	0.11 ± 0.04	Camera trap	J. Simonetti unpublishing data
Pteroptochos castaneus	12.2 ± 0.8	0.24 ± 0.4	9.00 ± 1.7	0.31 ± 0.05	0.05 ± 0.1	0.003 ± 0.003	Playback; transects	Ramírez-Collio et al., 2017
Pleuroderma thaul	6.16 ± 2.1	0.11 ± 0.03	4.33 ± 1.7	0.13 ± 0.03	0.05 ± 0.1	0.003 ± 0.003	Active count	Puente-Torres and Simonetti, 2016
Eupsophus septentrionalis	6.00 ± 1.5	0.13 ± 0.03	3.66 ± 1.8	0.10 ± 0.04	0	0	Active count	Puente-Torres and Simonetti, 2016
Ceroglossus chilensis	21.3 ± 8.7	0.30 ± 0.08	5.66 ± 1.3	0.20 ± 0.04	2.44 ± 0.7	0.28 ± 0.7	Pitfall trap	Russek et al. (2017)

with visual identification and incorporated in the land-cover type map. The updated landcover was rasterised to 25 m to have all the variables at the same spatial resolution. Maxent models were running using default settings and partitioning occurrences for training (70% of the total) and testing (30%) and pseudoabsences were randomly distributed across the study area. We run ten replicates by species and the best model were selected using the higher area under curve (AUC) training value. Final probability model of each species was transformed into a binary map using the equal training sensitivity plus specificity threshold criteria (Liu et al., 2013). Highest training AUC values, thresholds and variables percent contribution of each species Maxent models are presented in Supplementary material S1.

2.3.3. *Potential connectivity model*

Circuitscape 4.0 software was used to model the landscape resistance to movement of each species. This program calculates pairwise values of resistance to movement flow among local populations (focal fragments) based on all possible paths, not just the least-cost path, thus improving the estimation of flow over many generations (McRae et al., 2008). Habitat quality maps (conductance grid) for the seven-species, obtained from Maxent, were imported into Circuitscape to generate spatial resistance distances to the movement among sites. Suitability values were transformed by reversing their values because we need to connect low movement resistance with high quality areas (Dai et al., 2019). We run the analysis using a pairwise scenario where the same fragments which obtain species occurrences iterated with a focal node. The output for Circuitscape is a cumulative connectivity values map where higher values represent higher connectivity. To assess how similar species were regarding functional connectivity we assessed the spatial overlap in conductance cells between species using pairwise Schoener's D metric (Warren et al., 2010) and quantified the area of each land cover type used by each species to move, as predicted from the Circuitscape, connectivity model. Finally, to quantify and compare connectivity models between species we use the software CONEFOR SENSINODE 2.6 (Saura and Torné, 2009), we calculated the graph-based connectivity metric Index of Integral Connectivity (IIC) for each model using the centroids of focal fragments as nodes and the presence/absence of corridors as edges (links between nodes).

2.3.4. *Connectivity priorization*

We apply a spatial priorization analysis over potential connectivity maps obtained from Circuitscape analyses to establish species-specific corridors of movement. First, we use Zonation Software (Moilanen et al., 2014) to rank the study area based on their connectivity potential. Zonation produces a ranked landscape using a spatial optimisation process (Pouzols and Moilanen, 2014). The core-area priorization algorithm in zonation was selected to prioritize areas with the higher value of potential connectivity obtained from the sum of each raster generated for the seven species with Circuitscape software. Finally, Zonation output raster was used to estimate the least-cost network between the ranked landscape using the Cost Connectivity tool available in the Spatial Analyst extension of ArcGIS 10.4 Software (ESRI, 2017). The focus of the Cost Connectivity tool is defining the optimum network of least-cost paths rather than to separate paths connecting regions. On the resulting least-cost connectivity network, the species is assumed to move using the paths. Two scenarios of networks of least-cost paths were defined; the first one included a corridor network of the seven species. While the second scenario, comprised only the species under any category of threat (Vulnerable and Endangered), as stated in the red list of Chilean species (Chilean Ministry of Environment, MMA, 2018). For the last scenario, three species were selected, two vulnerable (*L. culpaeus* and *L. guigna*) and one endangered (*E. septentrionalis*). The two optimised corridor network scenarios were analysed by overlaying the updated land cover and the two-corridor connectivity models obtained with Zonation. To compare both scenarios, we calculated the total sum of network costs and average cost values using ArcGIS software (ESRI, 2017).

3. Results

3.1. *Species distribution models*

Species distribution maps showed that the probability of the presence of each species was related to the presence of more extensive and continuous fragments of native forest in areas with a more complex topography (Fig. 3). While almost all of the species present a similar distribution of suitability values, showing higher values in central areas of the study area, related to higher elevation and terrain complexity. *C. chinga* and *L. culpaeus* showed a different pattern, with suitability for *C. chinga* being more related with flat areas near the coast and in the case of *L. culpaeus* present higher values of suitability around of all study area (Fig. 3). Independent of the species, lowlands and areas near the coast dominated by plantations were identified as with the lower values of suitability. The binary model showed species differing in their land cover type preferences, where *L. culpaeus* and *C. chinga* occupy forest plantations and clear-cutting areas more than native vegetation, while *L. guigna, P. castaneus, E. septentrionalis* and *P. thaul* distributed mainly in native forests (Fig. 4). Also, *C. chilensis*, use frequently both native forest as plantations in similar proportions (Fig. 4).

3.2. *Potential connectivity model*

Connectivity models differed in the amount of area of the potential connectivity network, particularly for *P. castaneus, L. guigna* and the two-amphibian species (*E. septentrionalis* and *P. thaul*). For these two-frog species, the model connected close native forest fragments, thus differing from the rest of species for which areas with high connectivity are distributed along the landscape, including different kind of land-cover types (Fig. 5). These results are consistent with the IIC index, whose higher values were related to the species with a more significant ability to move across habitats with similar structural complexity such as *L. culpaeus*. On the other hand, the native forest specialist *L. guigna*, *P. castaneus*, *E. septentrionalis* and *P. thaul* had the lower values of IIC index (Fig. 5). Generalist species (*C. chilensis*, *L. culpaeus* and *C. chinga*) exhibited the highest values of Schoener's D metric of overlap while forest specialist (*E. septentrionales*, *P. thaul, L. guigna* and *P. castaneus*) also exhibited high overlap among them (Table 2).

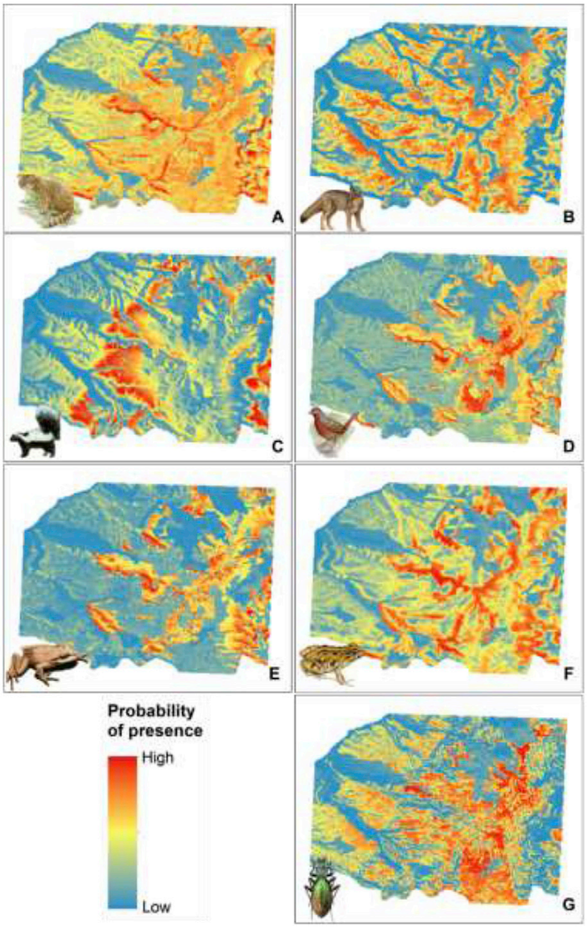

Fig. 3. MaxEnt species distribution maps for: A) *Leopardus guigna* (Threatened species); B) *Lycalopex culpaeus*; C) *Conepatus chinga*; D) *Pteroptochos castaneus*; E) *Eupsophus septentrionalis* (Threatened species); F) *Pleuroderma thaul* and G) *Ceroglossus chilensis* (Chilean hunting law).

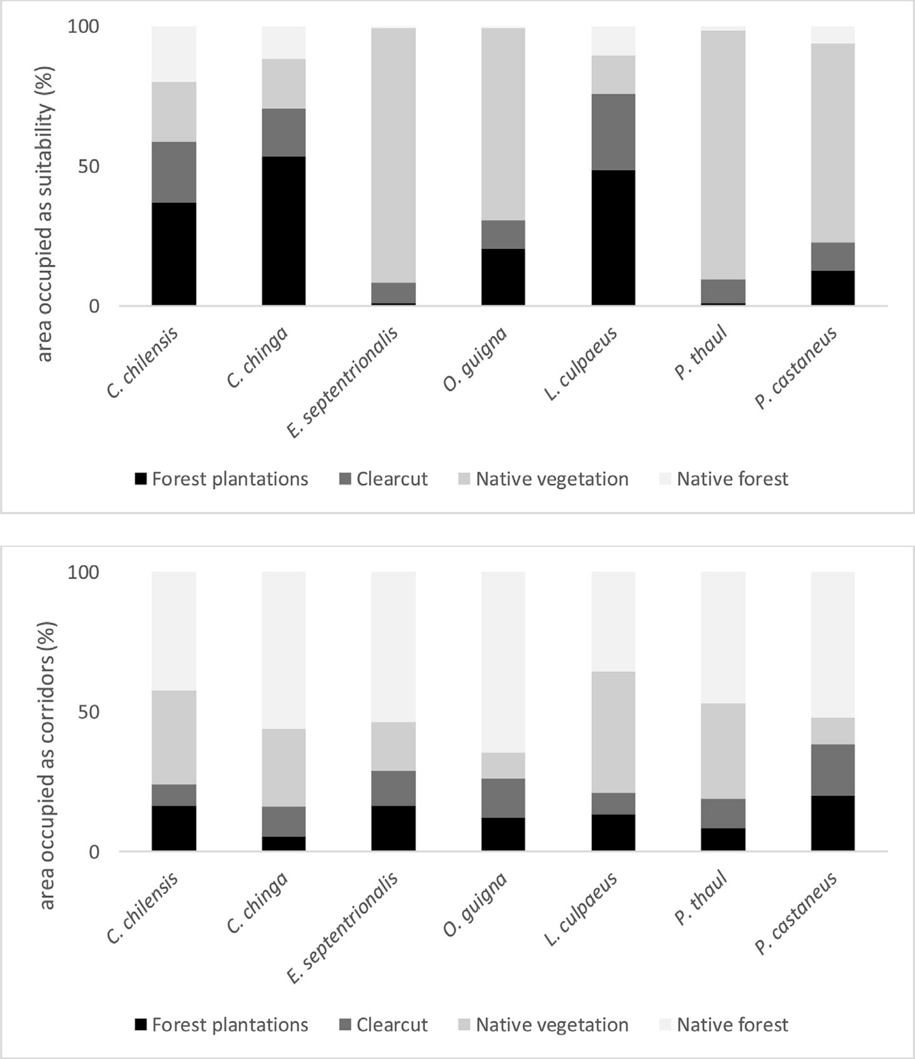

Fig. 4. The proportion of area occupied for each species analysed in the land-cover types presented in the study area, following the result of (a) species distribution models (final binary models) and (b) potential connectivity models.

3.3. *Connectivity priorization*

Spatial priorization based on Zonation identified as the most priority areas those found in the borders of the landscape, but also identified middle to high-value ranked fragments depending on the two corridor connectivity scenarios (Fig. 6). Least cost-path were different between scenarios (Table 3), the scenario that includes the seven studied species had a higher total cost in forest plantations areas and a higher average cost within the native forest, hence the movement of the species (final path) selected under this scenario was independent of the land-cover type. For the threatened species scenario, the total cost was higher in native forest areas, and the average cost was higher in natural areas. Hence, the movement behaviour of the species selected under this scenario was land-use dependent. The overall cost of the corridor network for the threatened species was higher than that for all species, thus indicating the degree of threat of a species is positively associated with the degree of landscape resistance to its movement.

4. Discussion

4.1. *Species distribution models*

Our results support that the type, topographic distribution, and amount of habitat patches for specialist animals could contribute to landscape connectivity. This contribution will be dependent on the level of habitat specialisation and the threat

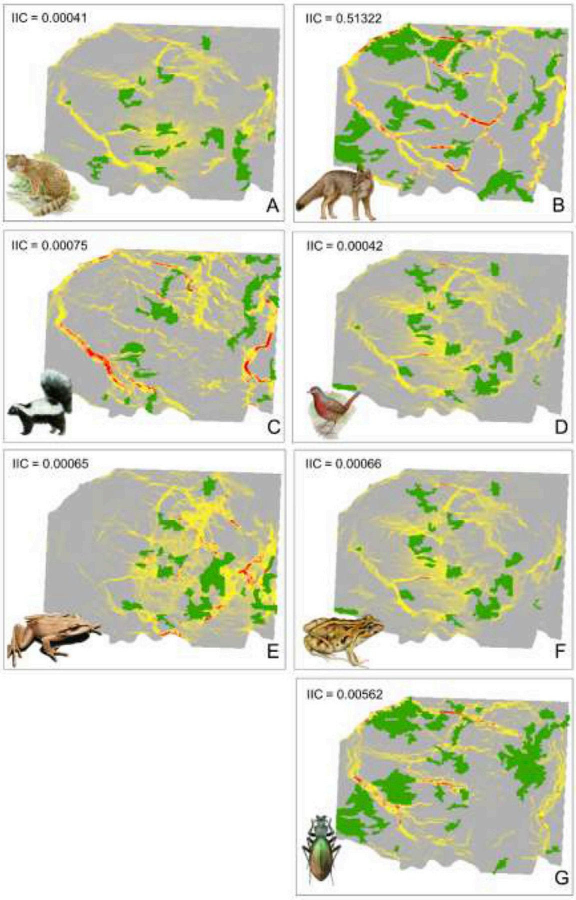

Fig. 5. Potential connectivity analysis for wildlife species living in an industrial forestry landscape, as based on least-cost modelling and circuit theory corridor implemented using Circuitscape (Focal fragments in green, yellow areas show low to high (red) current flow. A) *Leopardus guigna* (Threatened species); B) *Lycalopex culpaeus*; C) *Conepatus chinga*; D) *Pteroptochos castaneus*; E) *Eupsophus septentrionalis* (Threatened species); F) *Pleuroderma thaul* and G) *Ceroglossus chilensis* (Chilean hunting law). Upper left corner values shows Integral index of connectivity (IIC) from CONEFOR SENSINODE 2.6. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Table 2

Schoener's overlap index (higher values mean similar structural connectivity), between pairs of species for the top 10% of values in the connectivity model values made with Circuitscape. Higher values in bold.

Species	*C. chilensis*	*C. chinga*	*E. septentrionalis*	*L. guigna*	*L. culpaeus*	*P. thaul*	*P. castaneus*
C. chilensis		0.31	0.13	0.16	**0.63**	0.13	0.14
C. chinga			0.19	0.25	0.31	0.16	0.19
E. septentrionalis				0.40	0.13	**0.63**	**0.59**
L. guigna					0.15	0.51	**0.55**
L. culpaeus						0.11	0.14
P. thaul							0.51
P. castaneus							

of each species, as shown for individual species under different landscapes and movement context (Merrick and Koprowski, 2017). In fact, specialist species as *L. guigna* and *P. castaneus* are positively correlated for dense forest cover (>50%) provided by native fragments (approximately 70%), both in agricultural as forestry landscapes (Vergara and Simonetti, 2004; Gálvez et al., 2013; Moreira-Arce et al., 2016; Ramirez-Collio et al., 2017). Also, some species as *C. chilensis* can occupied plantations as native forest, but this beetle is more concentrated using the edge of native forest fragments as habitat and therefore, they could occupy this for movement or feeding (Barahona-Segovia et al., 2019). Our results also provided the first approximation for the habitat use on both amphibians associated strongly to the Maulino forest. On the other hand, generalist species such as *L. culpaeus* have been recorded occupying perturbed landscapes by forestry frequently (Moreira-Arce et al., 2016). In the case of *C. chinga*, our result support the generalist behaviour according to other studies (Donadio et al., 2004; Lantschner et al.,

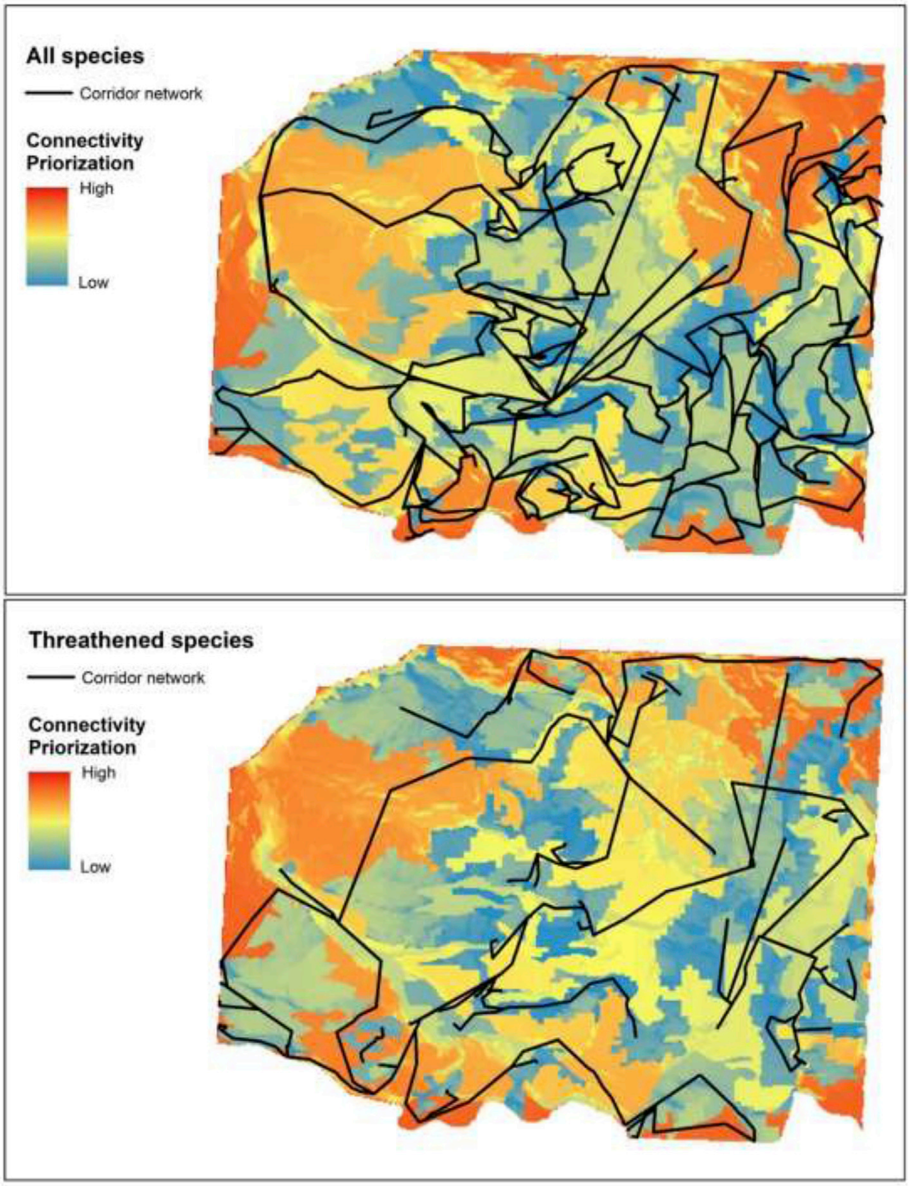

Fig. 6. Final corridors scenarios based on connectivity priorization with Zonation software. Black lines show the least cost path network across the landscape.

Table 3
The proportion of path cost of the least-cost network for the two final corridor scenarios.

Land Cover	All Species		Threatened species	
	Average Path Cost	Sum Path Cost	Average Path Cost	Sum Path Cost
Forest Plantations	4.3×10^{-5}	**0.02**	7.9×10^{-5}	0.03
Clear cutting	4.2×10^{-5}	0.02	8.3×10^{-5}	0.02
Native Forest	$\mathbf{4.4 \times 10^{-5}}$	0.02	7.5×10^{-5}	**0.03**
Native Vegetation	3.9×10^{-5}	0.01	$\mathbf{8.6 \times 10^{-5}}$	0.02

2012). Thus, availability of habitat must be essential in the landscape connectivity for specialist species, since they depend on other variables associated to native forest as prey abundance or physiological conditions (Delibes-Mateos et al., 2014; Barahona-Segovia et al., 2019).

4.2. Potential connectivity model

Dispersion is essential for animal population dynamics and to maintain the viability of them over time (Estrada et al., 2015; García-Valdés et al., 2015). However, specialist species depend on some landscape characteristics used as corridors. Our connectivity model results suggest that for the amphibian species their dispersion ability must be limited to close native fragment among them or depend on the structural complexity of matrix to move among fragments (Estades et al., 2012). Currently, for both amphibians species, dispersion capability is absent of the literature; however, we suspect, that dispersion in the landscapes must be realised at low temperatures (Iturra-Cid et al., 2014), which can be facilitated and decreased by high native forest cover, according to with our results (Fig. 4b; Barahona-Segovia et al., 2019). This information is critical for example, for planning conservation or predicts the distribution of frogs under climate change scenarios (Uribe-Rivera et al., 2017). Also, our connectivity model results for *L. guigna* support the idea that this small cat in fragmented landscapes increases the dispersion to avoid endogamy (Napolitano et al., 2015), using plantations as corridors. On the other hand, generalist species can move across landscape easily because not require specific habitat conditions, favoured for increase of edges or open areas, or because they have higher dispersion capability or biggest home range than specialist species (Donadio et al., 2004; Lantschner et al., 2012; Cerda et al., 2015; Moreira-Arce et al., 2016; Barahona-Segovia et al., 2019).

4.3. Connectivity prioritization

Threatened species have priority in conservation planning because its populations are decreasing due to habitat fragmentation, among other causes (see examples in Early and Thomas, 2007; Yumnam et al., 2014; Linnell and Leismeister, 2019). In our study, the threatened species group only need native forest fragments, which has middle to high conservation value and occupied less territory when we compared the connectivity prioritization with all species group. This information is relevant because not threatened species buffered the extinction risk of animals as *L. guigna*, *P. castaneus* or *E. septentrionalis*, giving them less priority in the connectivity model. Similar cases have been developed in other taxa, coinciding with the careful planning for connectivity prioritization and protect local threatened species (Early and Thomas, 2007). Although many species can be highly mobile in the landscape, others are less vagile and depend on certain key elements as old forest for the movement and promote dispersion across the matrix (Linnell and Lesmeister, 2019). Species with less information about movement rate as *E. septentrionalis* is a priority for new field studies and thus, improve potential corridors for less vagile species.

4.4. Forestry management in the anthropocene era

Through the multi-taxa methodological framework presented here, we have established different networks of connected habitat based on the landscape-optimised movement corridor for all-species and threatened species. We were able to identify potential corridors combining different current land uses and presented a network of corridors under different conservation scenarios. These results have relevance for forestry because they can achieve the Aichi targets in biodiversity matters. First, native forest fragments have middle-high conservation value and old plantations can be improved by structural complexity developing understory for specialist species that already occupy these stands for dispersion in the landscape (Napolitano et al., 2015; Ramírez-Collio et al., 2017). Second, although our proposed multi-taxa framework requires further evaluation of population viability, it provides preliminary evidence of networks of movement corridors assuring the persistence of species differing in their habitat preferences, sensitivity to native forest replacement and movement capacity. Forestry needs to incorporate biological corridors as a methodological task in their harvesting process and ensure that potential corridors that could be used in the connectivity network not being cleared. Thus, landscape elements having a more significant contribution to landscape connectivity and that, therefore, should be of priority for sustainable planning of forestry landscapes must be preserved. Our approach incorporates the dynamic of forestry landscapes through specifying the effects of clearcutting and plantations in a more advanced age on the landscape-scale movement of multiple species. The multi-taxa approach addressed in this study is often not adopted in landscape-level analysis, which typically focuses on the effect on focal species (Pitman et al., 2017). Finally, forestry in the socio-ecological framework is not well seen by the neighbouring communities; so, forestry could help to improve the population viability of threatened species by (1) habitat restoration; (2) connectivity in the landscape to avoid genetic consequences and detrimental consequences in these populations (McFadden and Dirzo, 2018). Another task could be addressed in environmental education to link forest sustainability with conservation biology and care and fortress of threatened and endemic species. This comprehensive package of measures can be applied to accomplish several Aichi targets. Our results provide a quantitative ecological basis for forestry guide and their management options towards the conservation of wildlife in industrial forestry landscapes.

Declaration of competing interest

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

Acknowledgments

We thanks to Alonso "Loncho" Salazar for animal pictures in Figs. 3 and 5. This work has been funded and supported by the Fondo Nacional de Desarrollo Científico y Tecnológico, Chile (FONDECYT) Nr. 1140657. PP thanks financial support of FONDECYT Nr. 1181677.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gecco.2020.e01108.

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