



Habitat loss of a rainforest specialist pollinator fly as an indicator of conservation status of the South American Temperate Rainforests

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Abstract

We estimate habitat loss and fragmentation in a hoverfly, *Aneriophora aureorufa*, used as a representative forest specialist species. This species is a pollinator specialist of two native trees, forming a triad endemic to the South American Temperate Rainforest (SATR). We combine species distribution models with species-specific requirements to estimate the habitat range of *A. aureorufa* over two non-overlapping time periods (before human settlement to 2000, and from 2000 to 2014). We analyzed the predicted distribution range of *A. aureorufa* in Chile, quantifying habitat loss in both periods and fragmentation in the latter. In addition, we evaluated the representativeness of the Chilean protected areas system in relation to the current habitat of the species. We found that the total habitat of *A. aureorufa* decreased by 68.3% compared to historic pre-settlement levels; in the period 2000–2014 the loss was 4.9%. The northern zone was the most affected by habitat loss and fragmentation, with an estimated total loss of 89.9% from the historic period to 2014, with the loss of 238.2 km² per year between 2000 and 2014. Eighteen percent of the habitat of *A. aureorufa* occurs within protected areas. We found an overrepresentation in the southern zone (24.79%) and an underrepresentation in the northern zone (3.44%). We propose that forest specialist species of the northern zone of the SATR could be threatened due to the high pressure of habitat loss and the underrepresentation of the Chilean protected areas systems.

Keywords *Aneriophora aureorufa* · Chilean temperate forests · Private and public protected areas · Protected areas representativeness · Syrphidae

Introduction

Habitat loss is one of the main causes of biodiversity loss worldwide (Vitousek et al. 1997; Pereira et al. 2010). Additionally, habitat fragmentation generates different ecological effects on the species (negative or even positive), however the effects on forest specialist species are usually negative

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(Tuff et al. 2016; Fahrig 2017). These processes affects almost all species, including key species such as decomposers, predators and pollinators, also generating serious impacts in ecosystem functions (Hooper et al. 2005; Quintero et al. 2009). Pollinators represent a group of significant importance because they support the reproductive process of almost 90% of angiosperms (Ollerton et al. 2011) and 70% of crop systems (Klein et al. 2007; Albrecht et al. 2012). Although bees and bumblebees are considered the main pollinators both in natural ecosystems and agroecosystems (Klein et al. 2007), other insects such as hoverflies and tachinid flies, make significant contributions to this process (Tan et al. 2002; Lander et al. 2009; Marshall 2012; Orford et al. 2015).

In forest ecosystems, flies are important floral visitors of native tree species (Klein et al. 2007; Ssymank et al. 2008; Lander et al. 2009). Among them, hoverflies can maintain long-distance gene flow of the plant that they pollinated, as has been showed in endangered trees from the South American Temperate Rainforest (SATR) (Ssymank et al. 2008; Lander et al. 2009, 2010). The high degradation of forest ecosystems worldwide (Hansen et al. 2013; Haddad et al. 2015), represents a serious threat to the flies' biodiversity, especially those that depend exclusively on these ecosystems (Rotheray and MacGowan 2000; González and Coscarón 2005). Previous studies have identified habitat loss and fragmentation as the major threats to Eristalinae hoverflies species (Rotheray et al. 2001). The strong dependence of hoverflies on one or two floral resources has been occasionally reported (Branquart and Hemptinne 2000); this is the case of the Eristalidae, *Aneriophora aureorufa* Philippi 1865 from the SATR, which makes it more sensitive to forest loss (Smith-Ramírez et al. 2014, 2016).

Aneriophora aureorufa, has been described as an endemic hoverfly from central Chile, which is a remnant of Gondwanic forests, is one of the biggest (body size) and most elusive species of flies of the SATR (Thompson 1999). The species has a mean body size of 20 mm (Shannon 1926); as a member of the Milesini tribe, it is highly probable that their larvae growths on tree cavities or dead wood (Thompson et al. 2010). The annual phenology shows that the northern adults are active between November and December, while Southern are mainly active in February (Barahona-Segovia unpublished data; Polidori et al. 2014). This species is dependent on only two main endemic floristic resources, *Eucryphia cordifolia* (ulmo, Eucryphiaceae) and secondarily *Myrceugenia planipes* (peta or petagua valdiviana, Myrtaceae). With the exception of the south, the SATR is actually distributed over small pieces of land, being seriously affected by deforestation, mainly due to replacement with exotic forest plantations, croplands, firewood extraction and fires (Mittermeier et al. 2004; Miranda et al. 2017). Chilean forests have been subject to some of the most intense

deforestation worldwide, while at the same time, having a clear deficit of protected surface in the zones most affected by this process (Armesto et al. 2005).

Given the high specialism of *A. aureorufa*, it is possible to develop models that show the effect of habitat loss and fragmentation at the landscape level based on one pollinator species. We propose to use this species as a study object that could be an indicator forest specialist species, highly sensitive to habitat loss and fragmentation.

In this study we aim to estimate (1) the habitat extent and environmental suitability of *A. aureorufa*, (2) the habitat loss of *A. aureorufa* over two periods (historic–2000 and 2000–2014), (3) habitat fragmentation of *A. aureorufa* over the period 2000–2014, and (4) the representativeness of *A. aureorufa* habitat in both Chilean Public and Private Protected Areas (PPA).

Methods

Study species

Eucryphia cordifolia and *M. planipes* are trees that are endemic to the SATR, reaching heights of around 40 or 20 m respectively, at the adult stage. *M. planipes* flowers during the austral summer in December and January, while *E. cordifolia* flowers from January to March. The flowers of both species are visited by *A. aureorufa*, mainly at canopy height (Smith-Ramírez et al. 2014, 2016). The flowers of both species are open and white and, to the human-eye, are some of the biggest white flowers of the SATR. While *A. aureorufa* depends on *E. cordifolia* and *M. planipes*, the pollination of these trees seems not to be dependent on this hoverfly (Smith-Ramírez et al. 2014).

Estimation of the habitat extent and suitability of *A. aureorufa*

Climatic requirements

We developed SDMs using the maximum entropy technique with MaxEnt software V3.3.3k, linking a set of environmental variables and an occurrence dataset (Phillips et al. 2006; Elith and Leathwick 2009; Elith et al. 2011). We used 19 bioclimatic variables of Pliscoff et al. (2014), which is homologous to the WorldClim project (Hijmans et al. 2005), but integrates a higher number of climatic stations for Chilean Temperate Rainforests, improving the quality of the SDM (Online Appendix S1). Our occurrence dataset was composed of records from eight entomological collections and our own records of occurrences collected during field work (1962–2017). Material examined belongs to the Museo Nacional de Historia Natural of Santiago (MNHNS), the

Museo de Entomología, Luis E. Peña (MEUC), the Instituto de entomología of the Universidad Metropolitana de Ciencias de la Educación (IEUMCE), the Museo de Zoología of the Universidad de Concepción (MZUC) and a collection pertaining to the Universidad Austral de Chile (UACH). In addition, we obtained occurrences of personal field collected points; personal observations performed by Barahona-Segovia, records compiled by us over 17 years in Chiloé Island, and records from a citizen science project Moscas Florícolas de Chile (<http://www.facebook.com/groups/774986852548819/>; <http://moscasfloricolasdechile.wordpress.com>), reaching a total of 50 occurrence records (Online Appendix S2). This number of records resulting from a wide search of source information, is associated with the rarity of this hoverfly. However, this value exceeds the minimum sample size to an accurate model considering the prevalence of the species and the background of modelling, according to van Proosdij et al. (2016) (prevalence < 5%, minimum of 17 occurrences).

First, we generated an exploratory SDM using all 19-bioclimatic variables, which calculates the importance of variables (percentage of contribution and permutation). Using R 3.2.2 software, we evaluated the normality of the data using the Shapiro–Wilk test and quantile–quantile plots (Royston 1982, 1983) (Online Appendix S3). Then, we determined the level of correlation between pairs of variables using the absolute correlation coefficient (Bradley 1985), generating correlograms (Online Appendix S4). We quantified the spatial autocorrelation of the occurrence points using Moran's index on a GIS (Brown 2014). We selected the variables with < 0.7 of correlation index and that showed a major contribution to the exploratory model. We ran the final model with a fivefold-cross validation technique using only the selected variables. The accuracy of the model was quantified using the area under the curve metric (AUC) of the receiver operating characteristic (ROC) and applying a partial least regression (PLS) to support MaxEnt results (Online Appendix S5 and S6). Finally, the probabilities under the 10-percentile threshold, (probabilities over 0.402) of the bioclimatic suitability model were excluded from the final model (Online Appendix S7). The response curves of the model were analyzed as a suitability response to bioclimatic variables (only for *A. aureorufa*) (Online Appendix S8).

Biotic interaction requirements

These requirements correspond to the biotic interaction of the hoverfly with the two floristic tree resources (Smith-Ramírez et al. 2014, 2016). We generated two SDMs, one for *E. cordifolia* and a second for *M. planipes* (Online Appendix S7), in order to identify the floristic resource for *A. aureorufa* in a spatially explicit way. The SDMs for these tree species follow the same modelling protocol

that we used for *A. aureorufa*. The occurrence tree records were taken from the Global Biodiversity Information Facility (GBIF) and the Herbarium of the Universidad de Concepción and consisted of 113 records for *M. planipes* and 151 for *E. cordifolia*. To estimate the offer of floristic resources in the landscape, we use the SDM of both trees, considering only the probabilities over percentile 10 as significant probability of presences (0.375 to *E. cordifolia* and 0.332 to *M. planipes*). As a result, we obtained a binary SDM of each tree species (presence–absence) (Online Appendix S7), which were summarized in a map of three categories: presence of only *M. planipes*, presence of only *E. cordifolia* and presence of both tree species (Fig. 1).

Habitat requirement

In previous pollinator studies *A. aureorufa* has only been recorded in fragments of primary native forest larger than ~ 10 ha, independent of their shape (Smith-Ramírez et al. 2014, 2016 and unpublished data). The habitat extent was estimated to three different dates: historic (pre-human settlement), 2000 and 2014 following the methodology proposed by Carvajal et al. (2018). To estimate the historic forest cover, we used the map of vegetation of Luebert and Plischoff (2006) and Lara et al. (2012), which estimates the forest extension according to bioclimatic conditions and pre-settlement period, respectively. We consider that this map is a proxy of the area covered by native forest before pre-human settlement.

We chose to map changes in forest area and fragmentation between 2000 and 2014, because the most comprehensive information available about forest changes was compiled by Hansen et al. (2013), who elaborated and published a complete map of worldwide forest distribution between 2000 and 2012, and subsequently amplified it to 2014 (<http://earthenginepartners.appspot.com/science-2013-global-forest>). As limitation, Hansen et al. (2013) product is unable to distinguish between native and exotic forests, and Chile has large extensions of exotic forest plantations. In view of this, we subtracted the areas covered by exotic plantations, which are registered by the national forest cadaster of Chile (CONAF-CONAMA-BIRF 1999), from the 2000 forest extent layer of Hansen et al. (2013). The outcome of this process is the native forest extent to the year 2000 and their losses, which were used in the habitat estimations.

The final habitat of *A. aureorufa* corresponded to patches of adult native forest (> 10 ha), with the presence of *E. cordifolia* and/or *M. planipes*, and with significant bioclimatic suitability (over 10-percentile) (Fig. 1).

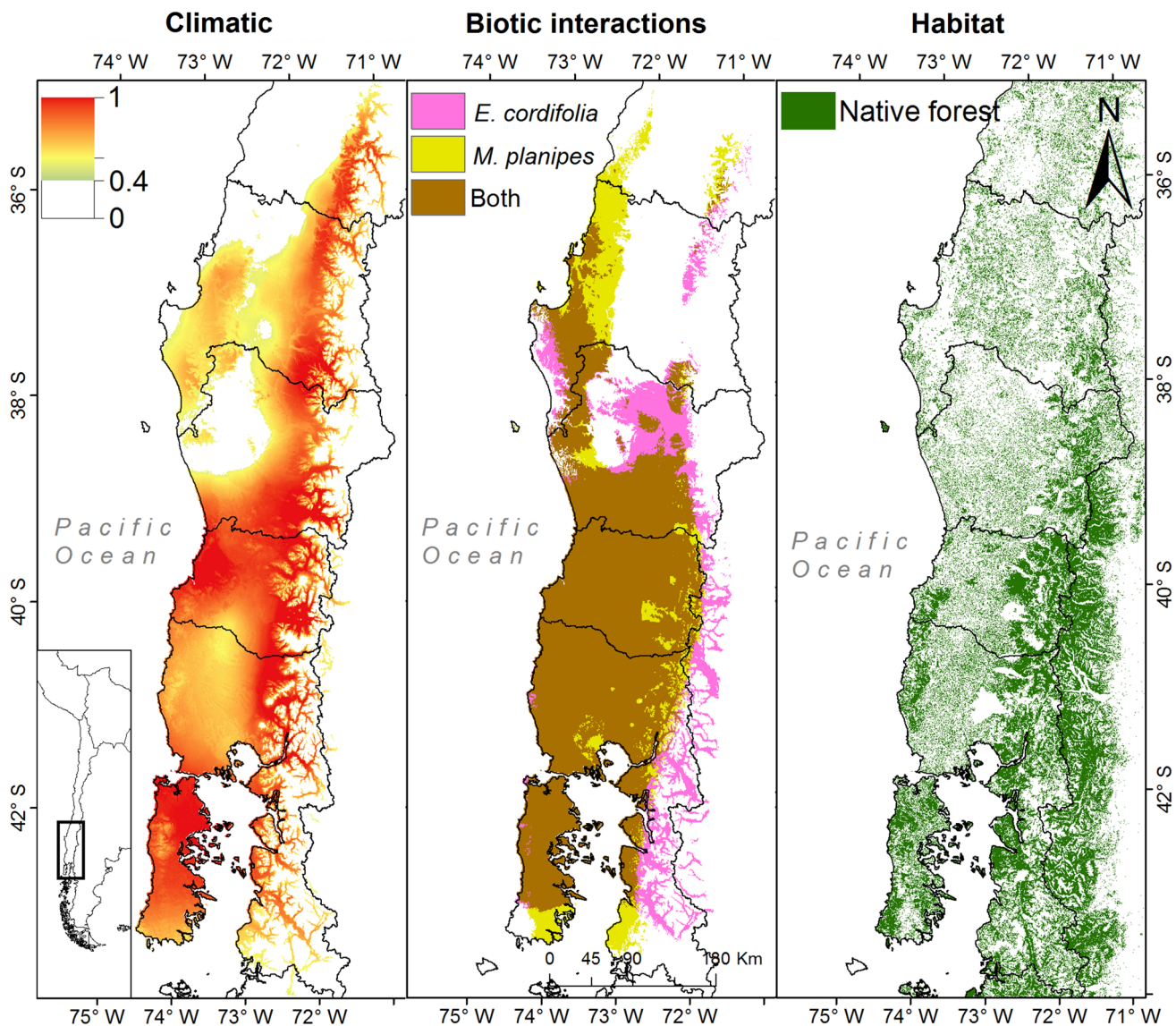


Fig. 1 Requirements of *A. aureorufa* model. Left: climatic requirement of Bioclimatic SDM of *A. aureorufa* considering only significant probabilities of presence (over percentile 10). Center: biotic interactions requirement imposed by the potential presence of rel-

evant floral resources. Right: habitat requirement (fragmentation) based on the presence of native adult forest with patches larger than 10 ha

Quantification of habitat loss and fragmentation

To quantify habitat loss, spatial patterns of change over the two study periods were studied using GIS. Habitat fragmentation was quantified only for the period 2000–2014 using data from Hansen et al. (2013) project. We used FRAGSTATS 4.2 (McGarigal et al. 1994) to quantify structural changes through the calculation of landscape metrics (mean shape index (MSI), edge density (ED), mean patch size (MPS), number of patches, and area of patches). First, we analyzed the changes in the complete habitat of *A. aureorufa*, and then after the model was generated, aiming to facilitate the interpretation of changes, the total habitat extent

was divided into three zones: north (35°S–38°S), center (38°S–41°S) and south (41°S–44°S) (Carvajal et al. 2018). We decided to not quantify fragmentation in the historic habitat extent, mainly because it is based on an estimation of the historical (Lara et al. 2012) and potential (Luebert and Plissock 2006) forest extent.

Assessment of protected areas representativeness

We overlapped *A. aureorufa* habitat in 2014 (current habitat) with a map of the Protected Areas of Chile, which considers National Parks (NP), National Reserves (NR), Natural Monuments (NM) and PPA (Sierralta et al. 2011). Subsequently,

we quantified the representativeness of *A. aureorufa* habitat in the protected areas by zone and type of protected area (north, center and south). Additionally, we compare the percentage represented in each zone with the overall percentage of representativeness by type of protected area.

Results

Habitat extent and suitability of *A. aureorufa*

According to our model, the historic extent of the habitat of *A. aureorufa* consisted of 76,411 km² which ranged

between 33.5°S and 43.5°S with a disjointed pattern between 33.5°S and 39°S (Fig. 2). This species was probably present in both Andean and Coastal ranges, and in part of the Chilean valley in Chile. From 39°S to 42°S its distribution was spatially continuous from Coastal to Andean ranges (74°W to 71.5°W).

The final SDM of *A. aureorufa* reached an AUC of 0.988 ± 0.002 , while the models of *E. cordifolia* and *M. planipes* reached an AUC of 0.984 ± 0.005 and 0.978 ± 0.022 , respectively (Online Appendix S5). All the variables were supported by PLS regression analysis, while the autocorrelation shows a random distribution of *A.*

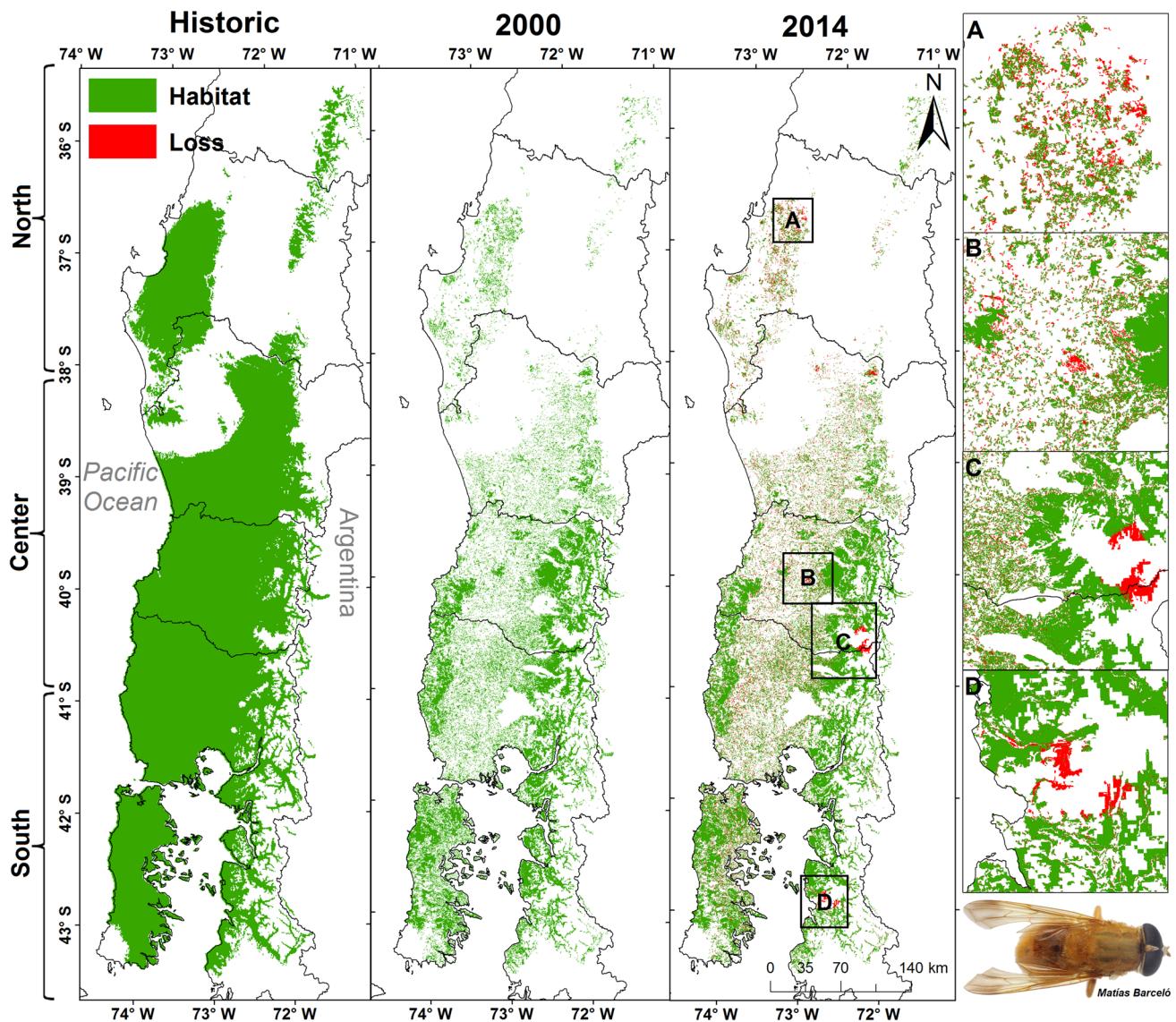


Fig. 2 Habitat extent for *A. aureorufa* in each period of time. Historic map of habitat on a pre-settlement period (left), map of habitat in 2000 (center) and 2014 (right). Habitat loss between 2000 and 2014 is represented in red. On the right insets of particular areas are

shown: (a) Cayumanqui hill in (72.7°W, 36.8°S) (b) relict *Nothofagus* forest near Los Lagos city (74.4°W, 39.9°S) (c) Caille volcano (71.9°W, 40.6°S) (d) Chaiten volcano (72.6°W, 42.8°S). On the extreme left, the different zones of analysis are specified

aureorufa occurrence points (Moran's Index = 0.485256; Z-score = 0.547247; p-value = 0.584209).

The variables that contributed most to the model were the precipitation of the coldest quarter (BIO19), the mean temperature of the wettest quarter (BIO8) and precipitation seasonality (BIO15) (Online Appendix S8). The percentage of contribution of each variable was 76%, 19.9% and 7.7%, respectively, while the permutation importance was 73.9%, 19.3% and 2.4%, respectively. The PLS regression supported the outputs of MaxEnt, ensuring the quality of the model (Fig. S4). The response curves of the more important variables show that suitability increases with the logistic pattern in relation to precipitation of the coldest quarter (BIO19), reaching a peak at 1400 mm. On the other hand, there is Gaussian behavior of the suitability curve in relation to mean temperature of the wettest quarter (BIO8), with a peak at 10 °C. Lastly, the behavior of the precipitation seasonality (BIO15) response curve increases steeply to a peak of 40 (coefficient of variation) (Online Appendix S8).

Habitat loss and fragmentation

Habitat loss from pre-settlement time to year 2014 was 68.3% (52.165 km²). The habitat extent in 2000 was 25.501 km², while in 2014 it was 24.246 km², which represents a loss of 4.9% compared with year 2000 (Fig. 2). The magnitude of loss from pre-settlement time to 2014 in the northern zone has been 89.9%, with this loss decreasing to the south. The recent losses show that the center zone had been most affected with a loss rate of 0.4% per year (50.1 km²/year), followed by the north with 1.49% (24.7 km²/year) and the south with 0.1% (14.9 km²/year) (Fig. 2).

With regard to habitat fragmentation, there was an increase in complexity, ED and number of patches between 2000 and 2014, while the MPS decreased (Table 1). The northern zone was the most affected, in terms of increasing complexity and ED (6.83% and 20.28%, respectively), while the number of patches did not change (0.08%). The southern zone was the less affected by habitat fragmentation, although complexity of patches and ED increased, with a slight increase in patch numbers (Table 1).

Protected areas representativeness

The current area of *A. aureorufa* habitat covered by protected areas is 4.359.8 km², which represents 17.96% of the total area. The southern zone has the largest proportion of protected habitat, corresponding to one quarter of this zone area (1.03% of the sum of the three areas), followed by the central zone, with 14.01% (41.14% of the sum of the three areas), and the northern zone with only 3.44% (57.79% of the sum of the three areas; Fig. 3). Overall, National

Table 1 Landscape metrics of *A. aureorufa*'s habitat

Zone	Metric	2000	2014	Change (%)
North	MSI	2.48	2.65	6.83
	ED	113.48	136.49	20.28
	MPS	63.56	50.19	-21.03
	NumP	2590	2592	0.08
	Area	1646.10	1300.91	-20.97
Center	MSI	2.54	2.64	3.98
	ED	77.36	81.84	5.79
	MPS	104.06	101.43	-2.52
	NumP	12,963	12,608	-2.74
	Area	13489.18	12788.81	-5.19
South	MSI	2.61	2.73	4.28
	ED	59.48	63.43	6.65
	MPS	211.70	207.22	-2.12
	NumP	4895	4900	0.10
	Area	10362.80	10153.69	-2.02
Total	MSI	2.55	2.66	4.43
	ED	72.38	77.03	6.42
	MPS	124.95	120.89	-3.25
	NumP	20,409	20,057	-1.72
	Area	25501.31	24246.66	-4.92

The table shows the landscape metrics of the total habitat and per zone of analysis. The metrics described are: mean shape index (MSI), edge density (ED, in m/ha), mean patch size (MPS, in km²), number of patches (NumP), area per habitat suitability level (area, in km²)

protected areas make a slightly greater contribution to protecting *A. aureorufa*'s habitat (54.40%) than PPA (Table 2). The protected areas with the highest proportion of habitat for *A. aureorufa* are Pumalin, a PPA (841.27 km²) and Vicente Pérez Rosales National Park (715.45 km²), jointly representing a total of 35.7% of protected habitat. Around 50% of the total protected surface area occurs within only ten protected areas (Table 3) distributed mainly in the southern zone. The representativeness by zone reveals that the north has an underrepresentation of -80.43%, while the southern zone is overrepresented (+38%) in relation to the total habitat (Table 2).

Discussion

Habitat extent and suitability of *A. aureorufa*

We found that the habitat of *A. aureorufa* is closely related to a temperate climate, and reflects the dependence of this species on the SATR (Di Castri and Hajeck 1976). The role of climate is highly important to insects, especially temperature, due to their ectothermic physiology (Huey et al. 2012). On the other hand, temperature and precipitation, also influence the phenology of the floristic resources of *A.*

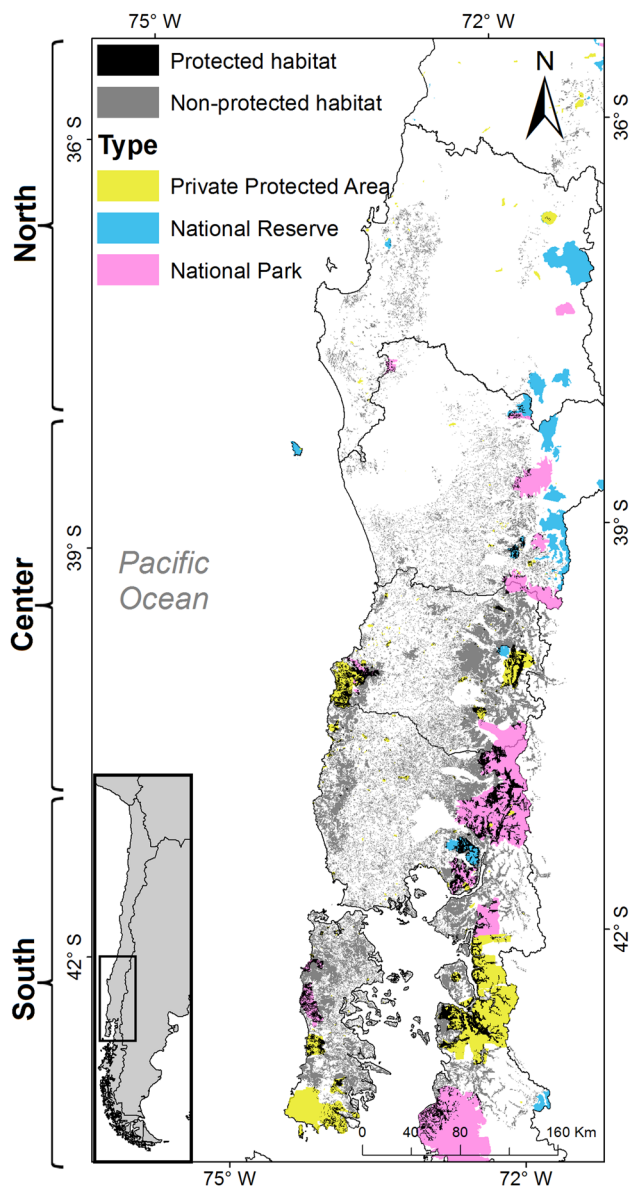


Fig. 3 Representativeness of protected areas with *A. aureorufa*'s habitat. In black, habitat of *A. aureorufa* within protected areas, in grey, habitat outside protected areas. The colors show the extent of the different types of protected areas. This figure does not include NM because they are too small and only represent 0.01% of the total habitat. (Color figure online)

aureorufa (Smith-Ramírez and Armesto 1994). *E. cordifolia* requires an abundant amount of water to ensure flowering, hence spring and summer precipitation should be a relevant factor to successful flowering. High precipitation in spring and summer has been considered a limitation on the abundance of the SATR pollinators, with the exception of Diptera (Smith-Ramírez et al. 2014).

It is important to highlight that the Chilean coastal range from 35°S to 39°S was a refuge zone to many taxa during last glacier maximum, which reached a maximum advance

18,000 years ago (Segovia et al. 2013). In this period the presence of glaciers inhibited the occupancy of the southern zone currently used by *A. aureorufa*, and the center and southern Andean range (Villagrán and Hinojosa 1997; Lara et al. 2012; Segovia et al. 2013). While we did not undertake specific analysis of habitat loss and fragmentation of the coastal zone inhabited by *A. aureorufa*, it is possible to observe significant habitat loss in maps reporting on coastal forests compiled by other authors (Smith-Ramírez et al. 2005a). As there is greater phylogenetic diversity in these forests (Smith-Ramírez et al. 2005b; Segovia et al. 2013) than in the Andean forests, then it is to be expected that genetic diversity of this population of *A. aureorufa* should be greater in this geographical zone.

Habitat loss and fragmentation

The spatial pattern of habitat loss follows a north–south tendency. The historic losses in the north were related to the replacement of native forests by crops (using fire to clear land) and exotic forest plantations during the period 1974–2000 (Miranda et al. 2017; Alaniz et al. 2016). Currently, loss is related to replacement by crops and mainly unintentional forest fires, as occurred in Cayumanqui hill in 2012, affecting 280 km² of coastal forests, forestry plantations, croplands and urban areas (Conaf 2014; Carvajal et al. 2018). In the Andean range of the central zone the current losses are mainly related to volcanism (Conaf 2014). However, in the coastal zone the influence of human activities associated with cropland expansion is significant (Miranda et al. 2017). Although the southern zone has been less affected by human induced disturbances, the eruption of the Chaiten volcano in 2008 caused significant forest loss (Echeverría et al. 2008; Carvajal et al. 2018). Furthermore, *M. planipes* and *E. cordifolia* are shade intolerant, slow-growing species (Donoso 2006) being *E. cordifolia* wood considered one of the most resistant woods to humidity from the SATR, and hence, one of the most appreciated trees used for firewood. This species were extensively logged across its distribution range from Chilean forests until recently, where conservation policies stopped their logging (Ramos-Jiliberto et al. 2009). Given this pressures, recovery from disturbances is more difficult especially because after cutting and subsequent re-sprouting, they are subject to grazing by livestock hindering their natural recovery (author personal obs.).

Fragmentation decreases core area and increases patch complexity, and ED could generate changes in core environmental conditions (Haddad et al. 2015; López-Barrera et al. 2007), particularly affecting specialist species of old-growth forests. Habitat fragmentation has been clearly identified in *A. aureorufa* habitat, especially in northern and central zones (Table 1), suggesting

Table 2 Representativeness of protected areas calculated in relation to habitat of *A. aureorufa* in 2014, per zone and type of protected area

Zone	National type	IUCN category	Area (km ²)	%	Zone/total (%)
North	NP	II	16.47	1.27	–85.26
	NR	IV	22.77	1.75	47.93
	NM	III	0.00	0.00	–100.00
	PPA	–	5.53	0.43	–94.81
	Total		44.77	3.44	–80.85
Center	NP	II	965.30	7.55	–12.11
	NR	IV	90.18	0.71	–40.40
	NM	III	0.09	0.00	–47.05
	PPA	–	736.73	5.76	–29.71
	Total		1792.30	14.02	–22.00
South	NP	II	1100.65	10.84	26.21
	NR	IV	171.94	1.69	43.11
	NM	III	0.22	0.00	72.11
	PPA	–	1244.91	12.26	49.60
	Total		2517.73	24.80	37.99
Total	NP	II	2082.42	8.59	100.00
	NR	IV	286.90	1.18	100.00
	NM	III	0.31	0.00	100.00
	PPA	–	1987.17	8.20	100.00
	Total		4356.80	17.97	100.00

The types of protected areas correspond to: National Park (NP), National Reserve (NR), Natural Monument (NM) and Private Protected Area (PPA). IUCN categories of protected areas are specified for reference. Numbers in columns represent the area in square kilometers (km²), percentage of protected area in relation to the total habitat of the zone (%), and a comparison between the representativeness of the types of protected areas in each zone in relation to the total representativeness expressed in percentage (zone/total)

Table 3 Contribution of the protected areas with more than 100 km² of *A. aureorufa* habitat

Protected area name	Type	Area (km ²)	%	Zone
Pumalín	PPA	841.27	3.47	S
Vicente Pérez Rosales	NP	715.44	2.95	S
Reserva Costera Valdiviana	PPA	303.88	1.25	C
Corcovado	NP	295.40	1.22	S
Chiloé	NP	250.55	1.03	S
Puyehue	NP	248.28	1.02	C
Reserva Ecológica Huilo Huilo	PPA	239.02	0.99	C
Alerce Andino	NP	211.58	0.87	S
Llanquihue	NR	171.81	0.71	S
Alerce Costero	NP	163.73	0.68	C
Parque Tantauco	PPA	138.55	0.57	S
Parque Tepuhueico	PPA	114.58	0.47	S

The type of protected area is described (for abbreviation please see Table 2), its extension, the percentage of representativeness (in relation to total habitat in 2014) and the zone in which they are located (south (S) center (C))

that core environments have been seriously degraded by edge effects. On the other hand, patch area has suffered a significant decrease, which could promote edge effects. Furthermore, the increase in edge and decrease in core

area produced by habitat loss could affect the viability of Dipteran larvae (Rotheray and MacGowan 2000; Rotheray et al. 2001). Core forest area is fundamental to the reproduction of this hoverfly, because their larvae probably have saproxylic behavior, similar to other Eristalinae (Barahona-Segovia personal Obs). This may be the reason why *A. aureorufa* has not been observed in isolated *E. cordifolia* or in small patches of this tree species. Edge area is associated with changes in decomposition rates as a result of the decrease in humidity and the increase in temperature, which affect the availability of decaying wood from edge to core area, thus decreasing the available habitat (Crockatt and Bebbler 2015). In addition to these effects on larvae habitat, the wind and temperature patterns in forest edges could complicate the use of resources and survival of this hoverfly species, due to the effects on performance and thermoregulation (Tuff et al. 2016). The estimated range of distribution of the species should be interpreted cautiously, because despite the wide distribution of *A. aureorufa*, it has a very low density and is highly rare.

The habitat loss and fragmentation patterns identified here are similar to other forest specialist species in the biome, as the case of the focal species *Scelorchilus rubecula* (Passeriform; Rhinocryptidae) which experienced a serious decrease in their habitat, reducing the amount of viable

populations particularly in the north zone of SATR (Carvajal et al. 2018).

Protected areas representativeness

The representativeness analysis showed a clear deficiency with regard to protection of the *A. aureorufa* habitat in the northern zone, which is supported by other forest taxa in SATR (Armesto et al. 1998; Plissock and Fuentes-Castillo 2011). Because of this, many studies have called for the Chilean Government to increase the protection of the northern distribution range of temperate forests, especially coastal forests (Smith-Ramírez et al. 2005a; Smith-Ramírez 2004). It is significant to note that the larger Chilean protected areas (private or not), including those which support the main habitat of *A. aureorufa*, contain several active volcanos, that increase the threats to any species within these areas.

Around 69% of the *A. aureorufa* habitat has been lost; while 82% of the remnant habitat is unprotected, and even the protected areas where this species occurs are threatened by active volcanism (the Caulle volcano is in Puyehue National Park) and forest fires (Conaf 2014; Carvajal et al. 2018). On the other hand, the use of pesticides in the edges between crops and native forests, threatens the permanence of hoverflies in general (Wratten et al. 2003). Although the use of pesticides has been an increasing concern in Chile, no studies have been carried out on the effects of pesticides on pollinators (<http://piensachile.com/2016/04/28058/>). In Chile there are around 100 species of hoverflies, most of which are endemic, such as *A. aureorufa* (Barahona-Segovia unpublished data). Threats to *A. aureorufa* could affect the main native pollinator of the SATR, the giant honeybee, *Bombus dahlbomii*. *A. aureorufa* has a Batesian mimicry with *B. dahlbomii*, sharing morphological, behavioral and trophic traits, e. g. the dependence on *E. cordifolia*, hence, both species could be threatened (Polidori et al. 2014; Barahona-Segovia personal obs.). We believe that this study may bring attention not only to the alarming situation of SATR pollinators, but also to other hoverfly species that may be under threat worldwide due to similar pressures (Vanbergen and Initiative the IP 2013).

Based on our results and those of other authors (Grez et al. 2006; Smith-Ramírez et al. 2007; Carvajal et al. 2018), we propose that forest specialist species in the northern zone of the SATR could be endangered, given that the northern area of this forest is reduced to fragments, and the representativeness of protected areas in this zone is very low. Also, worthy of note, the highest species richness of the SATR occurs precisely in the northern area of its distributional range (Armesto et al. 2005; Segovia et al. 2013), which merits the increased protection of this area, especially of the coastal range.

Finally, we believe that the methodology applied in this study shows that it is possible to combine SDMs with the specific requirements of the species to estimate niche extension, habitat loss and fragmentation. The spatially explicit dimension of this analysis enables us to assess the spatial representativeness of protected areas, and to identify priority zones and gaps of (un)representation, contributing to improve the management strategies and more effective designation of protected areas (UN United Nations 2010).

We determined that, overall, 68.3% of *A. aureorufa* habitat has been lost from its original historical distribution; in the northern zone of its distribution, the loss has been 89.9%, and fragmentation has also increased. Over the past few years of this century, increasing deforestation has also occurred in the central zone of SATR biome. Although there is constant threats and pressures to this biome that is difficult to control, the Chilean protected areas needs to be urgently enhanced, aiming to deal with lack of representativeness of biological diversity. Given this background, we believe that all forest specialist species of the South American Temperate Rain Forest could be threatened.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

Research involving human and animal participants This article does not contain any studies with human participants or animals performed by any of the authors.

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