The conservation and ecology of a newly described Amazon parrot; *Amazona lilacina*

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Thesis abstract

Parrots are one of the most threatened bird groups worldwide. The Ecuadorian Amazon parrot, *Amazona lilacina* was described as a full species in 2014 however, details on its ecology and status were missing. This research aimed to gather and collate information using natural and social science methods, to further our knowledge and provide an evidence base to inform conservation action. The five key areas of research were 1) conservation status, 2) species distribution, 3) fine-scale habitat preferences, 4) roosting dynamics, and 5) human interactions with the species. I conducted field surveys throughout coastal Ecuador to gather observations of *A. lilacina* and to locate communal roosting grounds. I conducted surveys of these roosting grounds and estimate the global population at 741-1,090 birds. Compared to data from 20 years ago, I suggest a population decline of 60%, meaning that *A. lilacina* fulfils the criteria for Critically Endangered on the IUCN Red List. I developed community questionnaires to record local peoples' observations of *A. lilacina* and used responses from >400 people, combined with my own observations, to model distribution. From this I predict that 17,772 km² of suitable habitat remains. Within this area, I assessed fine-scale forest characteristics over 35,800 m² and identified 36 tree species that *A. lilacina* uses. I developed a roost survey protocol to assess fluctuation in roost size and composition for one subpopulation. This method is often used to estimate the reproductive output of Amazon parrot populations, however my results from surveys over 36 consecutive months, suggest this is not possible and highlight a change in roosting dynamics since 20 years ago. I developed community interview questions, to understand local peoples' experiences and attitudes towards parrots, trapping and pet keeping, and results from >100 people suggest that parrot trapping and pet keeping occurs throughout the species' range. I developed a model to predict

the distribution of pet parrots, based on wild parrot abundance and human accessibility, and combined this with interview responses to assess the risk of parrot trapping. This highlighted that parrots in the south of the range are at greater risk, and I provide recommendations for development and continuation of support for conservation efforts with local communities.

Figure 1: The Ecuadorian Amazon parrot *Amazona lilacina* in a nest cavity in the Cerro Blanco Protected Forest, Guayas Province, Ecuador.

Chapter One: Introduction and aims

One million animal and plant species are threatened with extinction from human activities (Brondizio et al. 2019; Tollefson 2019). The current extinction rate is tens to hundreds of times higher than average for the past ten million years (Barnosky 2009; Ceballos et al. 2015; Brondizio et al. 2019). According to the IUCN Red List, of the 11,158 bird species described and evaluated, 14% are threatened with extinction (BirdLife International 2020a). This is a lower proportion than for other animals groups, perhaps because birds are more adaptable to environmental change due to their ability have large home ranges and travel long distances (Donald et al. 2010). Despite this, 200 bird species have become extinct during the last 400 years and this extinction rate shows no signs of slowing (Donald et al. 2010). In the last 20 years, the number of birds at the highest levels of extinction risk - listed as Endangered or Critically Endangered by the IUCN Red List - has increased from 503 to 684 (BirdLife International 2020a). Extinction rates for birds would be higher if it were not for successful conservation efforts. A review of Critically Endangered bird species with populations of <200 individuals and population declines of >80%, showed that over half would have become extinct between 1994 and 2004 in the absence of conservation interventions to mitigate specific threats (Butchart et al. 2006). Conservation interventions that prevented certain or a very high chance of extinction included control of exotic predators (e.g. rats and cats), intense captive breeding and reintroduction, nest management (nest guarding, clutch and brood fostering, provision of nest boxes), habitat protection (national park creation, land purchase and fencing) and habitat restoration (reduction of grazing or chemical use) (Butchart et al. 2006).

One of the largest and most threatened bird families is the parrots, Psittaciformes (Marsden & Royle 2015). They have a higher extinction risk than other comparable bird groups and 31% are categorised as threatened or extinct (Olah et al. 2016; BirdLife International 2020a). Over half of all parrot species are in decline due to anthropogenic threats, with the most frequent being agriculture, hunting, trapping, and logging (Olah et al. 2016). A recent study which assessed information for 192 Neotropical parrot populations found that since 2001, a declining trend was reported for 38%, and that information from which to assess population trend was missing for 32% (Berkunsky et al. 2017). The main threats were related to human activities, with 72% of populations threatened by agriculture, 68% effected by the pet trade, 55% threatened by logging, and 55% by human intrusion or disturbance (Berkunsky et al. 2017). The threat most closely associated with population decline, which is affecting 53% of populations, is capture for the local pet trade (Berkunsky et al. 2017). Worryingly, this study also found that conservation management, which could be in the form of just one of the following; nest box provision, nest and/or roost surveillance, habitat restoration, or improvement of natural cavities, was in place for less than 20% of all parrot populations (Berkunsky et al. 2017). Considering that 72% of populations are threatened by agriculture alone, conservation management for less than 20% of populations is insufficient to ensure parrots' survival.

Two of the bird species saved from extinction by conservation interventions between 1994 and 2004 were parrots; the Puerto Rican Amazon *Amazona vittata* and the Echo Parakeet *Psittacula eques* (Butchart et al. 2006). At the time *A. vittata* faced threats of habitat loss, hunting for food and pest control, trapping for cage-bird trade, and nest predation and competition, but during the ten-year period the use of artificial nest-sites, control of nest predators and competitors, captive breeding and reintroduction, and creation of protected areas, prevented it from becoming extinct (Butchart et al. 2006). *Psittacula eques* was threatened by habitat destruction and degradation owing to cyclones and introduced plants, introduced predators and food/nest-site competitors, but again extinction was prevented through extensive captive breeding and habitat management (Butchart et al. 2006). The Conservation Evidence website (conservationevidence.com) currently lists 17 actions that have been used to manage parrot populations. Unfortunately, 15 of these are considered to have unknown effectiveness based on limited evidence. The two actions that are suggested to be beneficial to parrot conservation are; the use of legislative regulation to protect wild populations, and translocation. One example shows that population densities of the Yellow-crested Cockatoo *Cacatua sulphurea citrinocristata* increased between 1992 and 2002 following a ban on trade in wild caught parrots (Cahill et al. 2006) and another showed that the number of parrot chicks taken from nests reduced after protective legislation in Africa, Asia and Australasia (Pain et al. 2006). Regarding parrot translocation for conservation, three studies of two programmes showed that translocation could result in successful establishment of populations and colonisation of new areas (Williams et al. 2020). The lack of evidence for the effectiveness of other actions listed, for example education programmes, community engagement, supplemental feeding, and artificial nest boxes, highlights the need for conservation interventions to be designed in a way that allows ongoing monitoring and evaluation.

The level of extinction risk to parrots varies depending on certain social factors and ecological characteristics (Olah et al. 2016). For instance, species whose range overlaps with countries where a greater proportion of the human population lives in urban areas, are at higher risk, and endemic parrots, or those restricted to a single country, are disproportionately likely to become extinct (Olah et al. 2016). Parrots with a larger body size and longer generation times, such as those in the genus *Amazona*, are also more likely to be classified as threatened (Forshaw & Knight 2010; Olah et al.

2016). Today 21 of the 36 species within this genus are listed as threatened or extinct in the wild (Olah et al. 2016; BirdLife International 2020a). As mentioned already, the threat most closely linked with population decline is capture for the local pet trade, which impacts some species more than others, because humans have a preference for which species they like to keep as pets (Berkunsky et al. 2017; Romero-Vidal et al. 2020). For example species within the genus *Amazona* are favoured due to their attractiveness and ability to mimic the human voice, and as a consequence they face increased demand and sell locally for a higher market value, factors which arguably increase their risk of extinction (Tella & Hiraldo 2014; Romero-Vidal et al. 2020). The method of capture can also affect the impact of trapping on a population. Results of 50 year simulations for Grey Parrot *Psittacus erithacus* populations, suggested that the inclusion of just a small number of adults in the capture 'harvest' had a far greater negative impact on the population than a similar number of juveniles (Valle et al. 2018). Furthermore, that irregular spikes in the number of birds trapped are more likely to be detrimental to populations than steady annual harvests, therefore if trade is going to continue, sustainably, it requires information and quotas regarding not only the number but also the demography of birds that are being trapped (Valle et al. 2018).

The study species for this thesis, *Amazona lilacina,* formally *A. autumnalis lilacina,* was brought to the attention of the European zoological community in the 1980's, when customs staff at European airports made large seizures of wild caught birds. In 1982 approximately 100 Amazons were confiscated in England, and in 1983 approximately 50 were confiscated in Germany (Pilgrim 2000). The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) is an international agreement set up to ensure that trade does not threaten the survival of species. Although *A. lilacina* is not specifically listed by CITES, due to a change in taxonomic nomenclature, all Psittaciformes have been included on CITES appendix II since 1981. Ecuador was one of the original countries to sign up to this agreement in 1975, furthermore in 1983 the country made the export of all indigenous wildlife illegal (Ecuadorian National Assembly 2017). Therefore, all export or import of wild caught *A. lilacina* is prohibited. Publicly available CITES documents report the scale of annual *A. lilacina* exports from Ecuador since 1981. Exports of live *A. lilacina* rose between 1981–1984, peaking at approximately 3,000 in 1984, before dropping off almost completely (CITES 1986). The vast majority of birds were exported to the USA who imported 2,780 birds in 1984 alone (CITES 1986). At this time, concern was raised about the extinction risk of *A. lilacina* (then *A. a. lilacina)* due to a significant population decline resulting from habitat destruction (CITES 1986). Further reports from the United Nations Environment Programme World Conservation Monitoring Centre show that between 1981 and 2009, 2,871 *A. lilacina* were imported by the USA with trade halting in 1994, 150 were imported by Japan in 1985, and 254 birds were imported by Europe with trade halting in 1988 (UNEP-WCMC 2011). At least 101 of the confiscated wild caught birds entering Europe were distributed between multiple zoological collections, including Chester Zoo, and went on to form the basis of the European Association of Zoos and Aquaria *Ex situ* Population (EEP) managed by Dr Mark Pilgrim since 1993 with assistance from myself since 2012 (Pilgrim 2000). The EEP currently consists of 62 birds distributed between 24 participating collections, and represents the only coordinated captive breeding programme for this species worldwide. It is however likely, that more birds exist in the private trade in Europe and the USA, and in rescue centres and zoological collections in Ecuador.

The taxonomy of *A. lilacina / A. a. lilacina* had been questioned for a number of years, and the arrival of these birds into Europe posed a unique opportunity to investigate this further. Originally, *A. lilacina* was deemed a sub-species or race of the *Amazona autumnalis* group, along with *A. autumnalis autumnalis*, *A. a. salvini*, and *A. a. diadema,* known collectively as Red-lored Amazons (Figure 1). The group's original distribution was believed to stem from the eastern slopes of Mexico, through eastern Nicaragua, Costa Rica, Venezuela, to western Colombia and western Ecuador, with *A. a. diadema* occurring in a geographically isolated population in north-western Brazil (Forshaw 1989). Investigation into the morphology, genetics and behaviour of the four different sub-species through detailed study of captive birds and museum specimens, revealed that *A. a. lilacina* was in fact distinct, and in 2014 it was classified as a full species by BirdLife International (Pilgrim 2010; del Hoyo & Collar 2014).

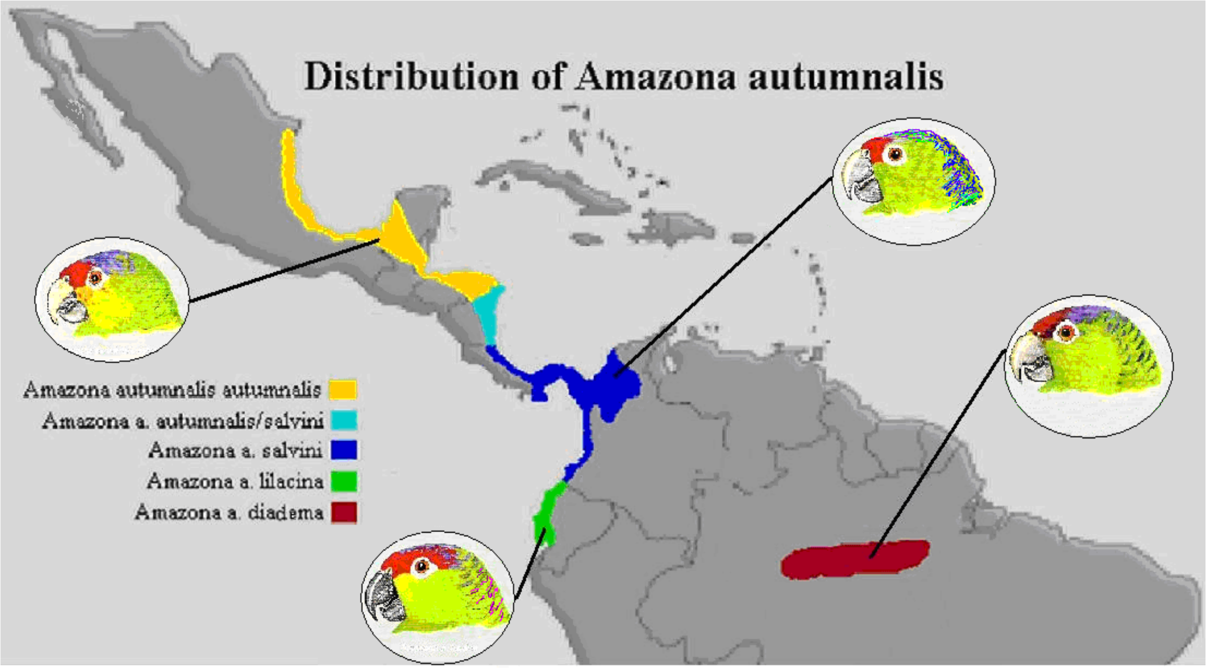


Figure 1: *A. a lilacina* was originally believed to be a sub-species or race of the *Amazona autumnalis* group along with *A. a. autumnalis*, *A. a. salvini*, and *A. a. diadema.* Figure courtesy of Dr Mark Pilgrim (Juniper & Parr 1998; Pilgrim 2010).

The reclassification of *A. lilacina* highlighted a need for research into the species *in situ* status, initially to determine the overall extinction risk. Whilst listed as part of the

A. autumnalis group, Red-lored Amazons had (and still have) a collective IUCN Red List categorisation of Least Concern (BirdLife International 2018a). However, it was already known that the *lilacina* sub-species was likely to have a very small population size, estimated at just 400-600 birds (Juniper & Parr 1998) and be threatened by deforestation and persecution (Ridgely & Greenfield 2001a). Despite this, detailed information on the range, distribution, preferred habitats, behaviour, and specific threats was lacking, making it hard to implement effective conservation measures. In 2012, prior to the start of my PhD research, Chester Zoo (R. Biddle and M. Pilgrim) began initial field investigations and communications with Ecuadorian NGOs. We conducted roadside surveys for species presence and pilot roost surveys at one well-known communal roost site. This information, alongside existing historical records of trapping and deforestation rates in coastal Ecuador, resulted in the species being listed as Endangered by the IUCN Red List in 2016 under criteria C2a(i); a global population of less than 2,500 mature individuals and no subpopulation estimated to contain more than 250 mature individuals (IUCN 2012; BirdLife International 2018b).

From initial field surveys and previous literature, *A. lilacina* is believed to occur in dry forest habitats, and be loyal to traditional roost sites that are used by most of the individuals occurring in the local area (Berg and Angel 2006, Kunz 1996, Ridgley and Greenfield 2001). However, the function of this communal roosting behaviour is unclear. It has been shown that other Amazon parrot communal roosts are ten times larger when food is more highly dispersed, suggesting that gathering at roost sites may help to inform foraging strategy (Salinas-Melgoza et al. 2013). Studies have shown for example that parrots are able to minimise intraspecific competition by choosing different departure flight paths to their roost mates (Chapman et al. 1989; Boyes & Perrin 2009). On the contrary, it is also suggested that when food resources are low, it is energetically favourable for parrots to avoid long displacements to the large roosts and find alternative shelters near foraging areas to spend the night (Seixas & Mourão 2018). Another common theory is that communal roosting provides protection from predators, for example that the flocking behaviour reduces the risk from avian predation (Enkerlin-Hoeflich et al. 2006) and the location of the roost site, for example on a mangrove island, provides protection from land-based predators (Kunz 1996). Either way, if suitable roosting habitat is limited, this is likely to influence overall population health. The study of communal roosts has also been used widely to monitor long-term trends in population size and structure, from which to infer information about a population's reproductive health. This is particularly useful for Amazons that are cryptic and secretive during the day, and therefore may be especially hard to observe when nesting (Collar 2000; Enkerlin-Hoeflich et al. 2006). In many cases roost size diminishes gradually as the breeding season advances when breeders roost in nests, and sharply increases after young fledge and follow their parents to the roost site (Dénes et al. 2018). For *A. lilacina*, it has been shown that parental responsibilities during the nesting period may explain fluctuations in the number of birds attending the communal roost site. These fluctuations can be used to estimate the number of reproductive birds within the population (Berg & Angel 2006). Therefore, the study of roosts, through roost surveys, can be extremely useful to understand roost function, estimate the size population, and measure reproductive health.

Zoological collections have an obligation to conduct research to inform conservation as per the European council directive of 1999, and one of four objectives listed for all EAZA members, is to conduct and fund *in situ* conservation. As an Endangered Ecuadorian endemic species, *A. lilacina,* is a high priority for research to inform conservation planning for its survival. Parrots are notoriously hard to study, they breed slowly, have wide ranges, may exhibit unpredictable and cryptic behaviour, and have been described as “calculated to defy the scientific investigator” (Collar 2000). Consequently, information on population parameters and ecology to support conservation status assessments is often missing (Berkunsky et al. 2017). Local communities in the rural coastal provinces of Ecuador are closely connected with *A. lilacina*, and it is essential that they are involved with research and conservation activities. Throughout this thesis, I refer to *A. lilacina* as the Ecuadorian Amazon, but acknowledge that it is also referred to in English as the Lilacine Amazon. Neither of these common names have Spanish translations that are used in Ecuador - most local communities refer to “loro frentirrojo” (Red-lored Amazon), which in English describes the *A. autumnalis* group and includes *A. a. salvini* in northern Ecuador. To avoid confusion, the name *A. lilacina* was used in all communication with local communities, and photographs were used to confirm identity. For consistent naming of all other parrot species, I follow the International Ornithologists Union (IOC) bird name nomenclature (Gill et al. 2021).

My goal for this research was to work in partnership with local communities to document a body of knowledge about *A. lilacina’s* ecology and threats, to inform its conservation. My five research aims, with specific objectives are listed below:

1) Review the conservation status of *A. lilacina* in order to evidence a comprehensive ICUN Red List status assessment:
   i. update the current known Extent of Occurrence and estimate area of daily dispersal;
   ii. estimate the global population size;
   iii. determine any change in roost size as an indicator of population trend;

iv. quantify the prevalence of pet parrots within the species' range.

2) Assess the global distribution of *A. lilacina* through field observations, community surveys and species distribution models:

   i. collate all known species locality records, including from our own observations, reports from expert ornithologists, and reliable eBird records (2010-2020) and use these to build distribution models;

   ii. collect data on local peoples' experiences and observations of wild *A. lilacina* through structured face-to-face interviews;

   iii. assess community survey data based on different quality filters and use these data to build distribution models;

   iv. determine the best performing distribution models built from species records and community reports, and compare their outputs in order to direct future field investigation.

3) Describe the fine scale habitat characteristics in areas used by *A. lilacina*:

   i. confirm which tree species are required for feeding, nesting and roosting;
   ii. compare the habitat characteristics between areas used and not used by *A. lilacina* to distinguish key characteristics associated with presence;
   iii. predict the suitability of habitat in an area in which *A. lilacina* are believed to occur but species observations were not collected during this study.

4) Assess the attendance and composition of *A. lilacina* at a well-known roost site to understand roosting dynamics;
    i. estimate the size of the subpopulation from the maximum number of birds observed during roost surveys;
    ii. determine if there is a difference between the number of birds observed during morning and afternoon surveys, and which is likely to give a more reliable representation of roost size and seasonable fluctuation;
    iii. assess whether roost surveys can be used as an indicator of population reproductive health.

5) Identify areas where *A. lilacina* are most at risk from trapping and pet keeping, and understand the frequency and drivers of these behaviours, to inform conservation action planning;
    i. conduct surveys across coastal Ecuador to locate communities with pet parrots;
    ii. use the locations of known pet parrots, to predict the distribution of pets throughout the species range, using variables related to parrot availability, opportunity and demand;
    iii. interview local people from communities where pet parrots are present, to quantify the level of parrot ownership, trapping and the attitudes towards these behaviours;
    iv. develop a trapping pressure index based on model predictions, locally reported incidence and attitudes towards parrot capture and ownership, to highlight areas of possible increased extinction risk.

Chapter Two: Conservation status of the recently described *Amazona lilacina.*

Biddle R, Solis-Ponce I, Cun P, Tollington S, Jones M, Marsden S, Devenish C, Horstman E, Berg K, Pilgrim M. 2020. Conservation status of the recently described Ecuadorian Amazon parrot *Amazona lilacina*. Bird Conserv Int **30**:1–13.

Abstract

Amazona lilacina is a threatened species endemic to Ecuador, existing across a patchwork of mangroves, lowland coastal forests, agricultural and community owned land. The species was described in 2014 and listed as Endangered by the IUCN Red List, however, full assessment of the population was lacking. Using a combination of field observations, roost surveys and community questionnaires, conducted over the last twenty years, we provide up to date information on the species' Extent of Occurrence, estimate its global population size, and evaluate its level of threat. Our results suggest the species occurs across an area of 19,890 km² in three distinct geographically isolated subpopulations. Roost surveys across the range estimate the minimum remaining population at 741 - 1,090 individuals and we present evidence to suggest a 60% decline over the past 19 years in one part of the species' range. We conducted community questionnaires with 427 people from 52 communities. The presence of pet parrots was reported in 37 communities, including 17 communities who reported pet *A. lilacina*. From this we predict that over half of all communities within our study area keep parrots as pets and at least 96 communities keep *A. lilacina*. Our findings justify an IUCN Red Listing of at least Endangered for this species and highlight need for conservation support. In order to assess population health in more detail, further research is required to assess genetic diversity and roost dynamics, and

to identify areas that may be important for feeding and nesting throughout the range. As many of these areas are likely to overlap with community owned land, we suggest that future conservation actions should revolve around, and be led by these communities.

Introduction

A third of all Psittaciformes are classified as threatened and over half of all populations are in decline, yet population parameters to support conservation status assessments are missing for many neotropical parrots (Berkunsky et al. 2017). Parrots are commonly threatened by habitat loss, persecution, and the pet trade, and many species are in need of conservation support (Berkunsky et al. 2017). Endemic parrots with small fragmented populations, and those such as amazon parrots with large body size and long generation times, are disproportionately at greater risk of extinction (Snyder et al. 1987; Purvis et al. 2000; Grady et al. 2004; Olah et al. 2016). Indeed 58% of species in the genus *Amazona* are currently listed by the IUCN as threatened or extinct in the wild (BirdLife International 2020a).

Amazona lilacina is endemic to Ecuador and was described as a full species in 2014 (Pilgrim 2010; del Hoyo & Collar 2014). An initial Red List assessment categorised it as Endangered due to its small and fragmented population (BirdLife International 2018b) however, detailed status information was lacking, uncertain or outdated. For example, the northern-most limit of the species' Extent of Occurrence (EoO), was historically recorded as southwest Colombia (Juniper & Parr 1998; Forshaw & Knight 2010), which is now believed to be incorrect. Additionally, its dispersal area and habitat preference was recorded as regions encompassing both mangrove and lowland coastal forest habitats (Ridgely & Greenfield 2001a; Athanas & Greenfield 2016), yet recent study confirmed the presence of a large roost in a non-mangrove habitat (Blanco et al. 2016).

Evidence suggests that population size and trajectory are strongly correlated with extinction risk among vertebrates (Grady et al. 2004) but, since *A. lilacina* was described as a full species, the population size has not been estimated and little is known about its trend in recent years. The species was reported to have undergone severe population decline prior to the mid-1980s in response to ongoing habitat loss and trapping pressure (CITES 1986; Ridgely & Greenfield 2001a), and by 1998 the population was estimated at just 400-600 individuals (Juniper & Parr 1998). However, this estimate is now almost twenty years old and its reliability is questioned due both to possible declines and to the recent identification of new roosts (*pers. obs.,* Blanco et al. 2016). Roost surveys have been used to estimate global and local population sizes in many parrots species (Gnam & Burchsted 1991; Martuscelli 1995; Matuzak & Brightsmith 2007; Dénes et al. 2018) and provide a tool for long term population monitoring (Wermundsen 1998; Wright et al. 2019). *Amazona lilacina's* communal roosting behaviour thus allows us to update the population estimate and conduct long term monitoring to assess population trajectory.

In response to the 'uplisting' of this species to Endangered in 2014, we re-examined its Red List status through personal field observations and collation of information from local experts, NGOs, and communities, over a seven year period to fulfil four objectives:

i. update the current known Extent of Occurrence and estimate area of daily dispersal;
ii. estimate global population size;

iii. determine change in roost size as an indicator of overall population trend;

iv. quantify prevalence of pet parrots within the species' range.

Methods

Study area and roost sites

Amazona lilacina is reliant on lowland coastal forests (Ridgely & Greenfield 2001a) where it feeds on a variety of fruits and seeds, and nests in cavities formed in the trunks and branches of tree species such as Pigío *Cavanillesia platanifolia* and Ceibo *Ceiba trichistandra* (Kunz 1996; Berg & Angel 2006). Although we know little about this species' reproductive behaviour, adults appear to explore cavities in October/November and produce one or two chicks that fledge between mid-February and late-March (Kunz 1996; Berg & Angel 2006). As with several other *Amazona* species, with the exception of breeding birds, or at least females during the incubation and early chick stages, it returns to communal roost sites every evening (e.g. *A. brasiliensis* (Cougill & Marsden 2004), *A. auropalliata auropalliata* (Matuzak & Brightsmith 2007), *A. amazonica* (de Moura et al. 2012)). For *A. lilacina,* these roost sites mainly occur on mangrove islands where birds join together every night (Berg & Angel 2006). Birds tend to arrive at sunset, flying in loose-knit flocks made of paired birds, single birds, triplets or small groups, often making loud contact calls as they fly. For this reason roost locations are often well known by local communities, who hear the birds as they arrive and depart the following morning. In contrast, during the day, birds are secretive and extremely difficult to locate as they feed silently and high in the canopy in small groups (Ridgely & Greenfield 2001a).

For this study, we identified four roost sites that are occupied throughout the year. We believe they contain a large proportion, if not all, of the remaining global population of this species and they are separated from each other by at least 50 km (Figure 1).

Roost 1 is located on a mangrove island in Manab*í* Province and was brought to our attention by Fundación Jocototo in 2012. Roost 2 is located in Santa Elena Province and is known to us through the work of Guillermo Blanco and José Tella (Blanco et al. 2016). This is the only roost we know of that does not occur in mangroves. Instead, the birds roost in stands of the locally known ‘algarrobo’ tree *Prosopis julifora*. Roost 3 is perhaps the most well-known roost, located northwest of the Gulf of Guayaquil, in the El Salado Mangrove Reserve where mangrove islands have been frequented by *A. lilacina* since at least the early 1990s (Berg & Angel 2006). Roost 4 is situated southeast of the Gulf of Guayaquil on an island within the Manglares Churute Ecological Reserve. It was located in 2016 through our community questionnaires.

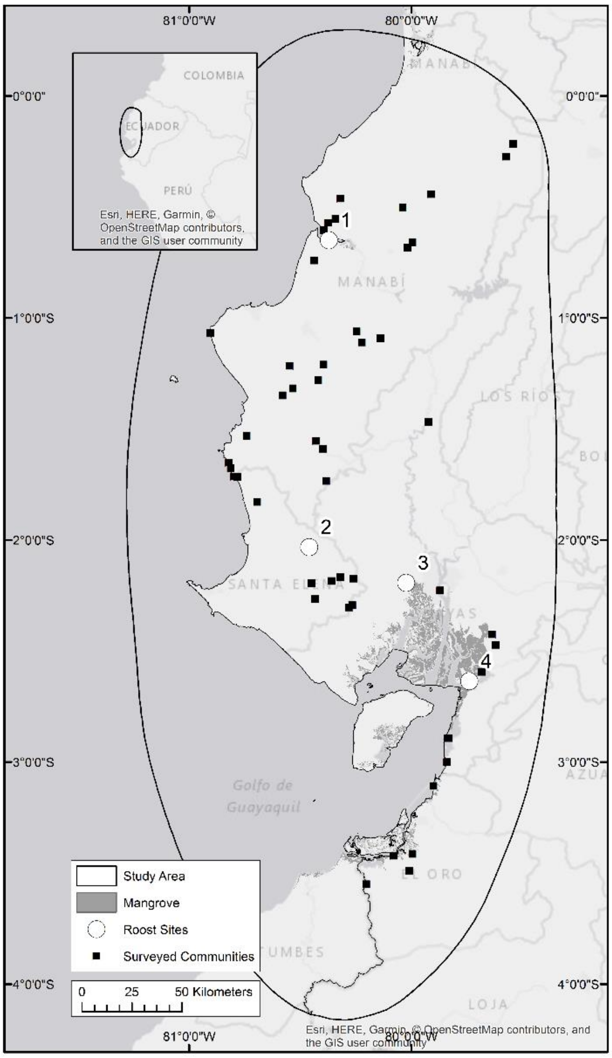


Figure 1: Four *A. lilacina* roosts are believed to contain the majority of the global population, three of these occur on mangrove islands. Mangroves (Hamilton & Casey 2016) and communities taking part in researcher led questionnaires are indicated.

Field observations

Observational data collected during ten field trips were used to address Objective i (November 2012, January and August 2014, November 2015, August 2016, January and March 2017, February 2018, January and August 2019). Field trips lasted two to three weeks during which we investigated potential areas of suitable habitat, verified any recorded sightings of individuals, and monitored known and newly reported roosts. Data collection was informed by: 1) existing information on known distribution and habitat use (Juniper & Parr 1998; Ridgely & Greenfield 2001a; Berg & Angel 2006; Forshaw & Knight 2010; Athanas & Greenfield 2016); 2) information on habitat distribution from Google Earth and available ecosystem maps (Ministerio del Ambiente 2012); 3) direct communication with local NGOs, ornithologists, local guides and bird tour companies and 4) communication through researcher-led questionnaires with local communities.

All sightings of perched *A. lilacina* made by ourselves, Fundación Jambeli staff, and Juan Freile within the last ten years were georeferenced (sightings of birds in flight were omitted). eBird presence data were lacking, however complete checklists that failed to report *A. lilacina* were used to gain an idea of absence areas: a total of 34,974 complete checklists for mainland Ecuador were downloaded in February 2019.

Roost surveys

To meet Objective ii, we conducted repeat surveys at all roosts. Although these were not located through systematic survey, they represent the combined current knowledge regarding this species according to the authors, local experts and communities. Initially, we conducted practice censuses at each roost to identify the best vantage points. Surveys were then conducted twice per day and where possible, for a minimum of four days to control for intrinsic variability (minimum of two, maximum of 20 consecutive surveys). To maximise our chances of counting all individuals leaving or arriving at each roost, morning surveys began before sunrise and lasted for two hours, whilst evening surveys began an hour before sunset and finished when it was too dark for birds to be identified. To reduce observer bias, all surveys were carried out by a combination of the same three researchers (RB, IS, PC), with one person counting and identifying birds using binoculars, the other keeping record. Roost sites are separated by at least 50 km and it has been suggested for other Amazon species that if roosts sites are isolated by >8 km, daily movement between roosts is unlikely (Cougill & Marsden 2004). Still, to account for possible movement of birds between roosts, which could result in counting the same birds twice, only roost surveys conducted during the same weeks of each year were used to estimate population size. Unfortunately, Roost 1 was disrupted and not occupied by Amazons during one year of the study, thus an average of counts before and after this disruption, but prior to the next global count, was used. The sum of these counts is presented as an estimated range in minimum global population size during the given time frame. Counts conducted in March are likely to include both adult and juvenile birds returning to the roost after the breeding season, so are suggested to be the most inclusive estimate.

Surveys from 2014 onwards at Roost 3 were conducted from an observation tower within the town of Puerto Hondo, approximately 300 m in front of the roost, allowing a full view of each parrots' flight path to and from the roost. This tower gives a good view of the roost area and approximately 1.2 km on either side. Morning surveys were conducted, by the same researcher, who attended the vantage point from 05h30–

07h30. At this roost birds are only seen flying in one direction (into or out of the roost) and therefore it is unlikely that birds were double counted. A consecutive day counting regime was used - the last four days of each month, which has been previously found to be more precise than counting on random days throughout the month; the regime used in 1999/2000 (Cougill & Marsden 2004; Berg & Angel 2006).

To facilitate Objective iii, we compiled all available surveys conducted at Roost 3 to assess long-term change in the size of this roost over time. Survey data were available from June 1999–May 2000 (conducted by Berg and Angel 2006) and for various months between November 2015 and May 2018.

Community questionnaire

To address Objective iv, information on the presence of pet parrots was gathered through researcher-led questionnaires in 52 communities within the study area (Figure 1). A total of 427 people took part, representing between 4 and 23 households per community. 'Open Street Map' (OSM) was used to categorise communities as hamlets, villages, or towns. Communities were selected due to their close proximity to lowland dry tropical forests (Ministerio del Ambiente 2012). Following trial surveys, questionnaires were carried out from January to July 2017. A combination of photographs, questions and sound recordings were used to ascertain if the participant could correctly identify *A. lilacina.* Participants were then asked: *"Are there any pet parrots in your village?"* and *"Which parrot species are kept as pets?"*

Questionnaires were conducted in Spanish and only the researcher (IS) and participant were present. Due to potential bias in self-reporting behaviour using direct questioning, especially in cases where that behaviour is illegal (Fisher 1993; Nuno & St. John 2014), we only asked participants to report the presence or absence of pets in their community as a whole. Participants could decline from contributing and were asked for verbal consent prior to participation once the purpose of the research was explained. Interviews were anonymous and data were coded to ensure that no individuals could be identified.

Data analysis

For Objective i, observation locations were used to estimate the Extent of Occurrence (EoO) using the IUCN Red List guidelines (IUCN 2012). ArcGIS was used to calculate the EoO, defined as "the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of current occurrence of a taxon" (IUCN 2012). The minimum bounding convex polygon tool was used within ArcToolbox to estimate area of EoO, with no exclusion areas.

To estimate the area of land that birds are likely to disperse over daily, buffers of 10 km were created around observation points; this is suggested to be the approximate diurnal ranging area of *A. auropalliata* in Costa Rica (Salinas-Melgoza et al. 2013). Buffers were dissolved in ArcToolbox. To analyse possible movement between daily dispersal areas, absence points were created using eBird complete checklists that did not record the species. Data were filtered and extracted using the *auk* package in R and following suggestions on best practice from Johnston et al. (2019), by restricting checklists to <5 h duration, <5 km in length, and with <11 observers.

For Objective iii, count data from roost surveys conducted using comparable methodology were analysed to assess any change in the size of Roost 3 from 1999/2000 and 2017/2018. For this analysis, only morning counts were used owing to the conclusions of Berg and Angel (2006) who found that their morning counts were more consistent, larger, and thus more accurate. Additionally, Cougill and Marsden (2004) showed morning counts to be more precise for estimating size of other amazon roosts. A generalised linear mixed model (GLMM) with a Poisson distribution and ‘month’ a random effect was fitted to compare counts from the two data sets. All statistical analyses were conducted in R version 3.6.0 (R Core Team 2019).

For Objective iv, ‘Open Street Map’ (OSM) was used to identify all communities in the study area, in the categories of hamlet/village/town. Predictor variables were calculated for each community (surveyed and not surveyed) using the Euclidean Distance and Values to Points tools in ArcToolbox. These related to species availability, accessibility and land protection status: distance to nearest sighting/roost, elevation (Jarvis et al. 2008), distance to nearest road (defined by OSM), and inclusion status within the National System of Forest and Protected Vegetation 2015 (Ministerio del Ambiente 2012). Additionally, mean Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product, MOD13A3, averaged across the period 2010-2015, was included as a proxy of vegetation cover. Random forests were used to classify surveyed villages with and without pet parrots, and with/without pet *A. lilacina* (Breiman 2001). The predict function in this package was then used to predict the likelihood of pet parrots and pet amazons being present in the remaining non-surveyed communities within the study area. Communities with a predicted vote score of 0.6 or over, thus a greater than 60% probability were considered as likely to have pets.

Results

A total of 132 occurrence points were gathered, and analysis of eBird checklists resulted in confirmation of 4,626 points of species absence (Figure 2a). The estimated Extent of Occurrence is 19,890 km² within which 5,313 km² is used by the species during daily dispersal. According to the IUCN’s definition of subpopulations (IUCN 2012) we suggest that *Amazona lilacina* occurs in at least three distinct subpopulations separated by a minimum of 40 km (Figure 2b).

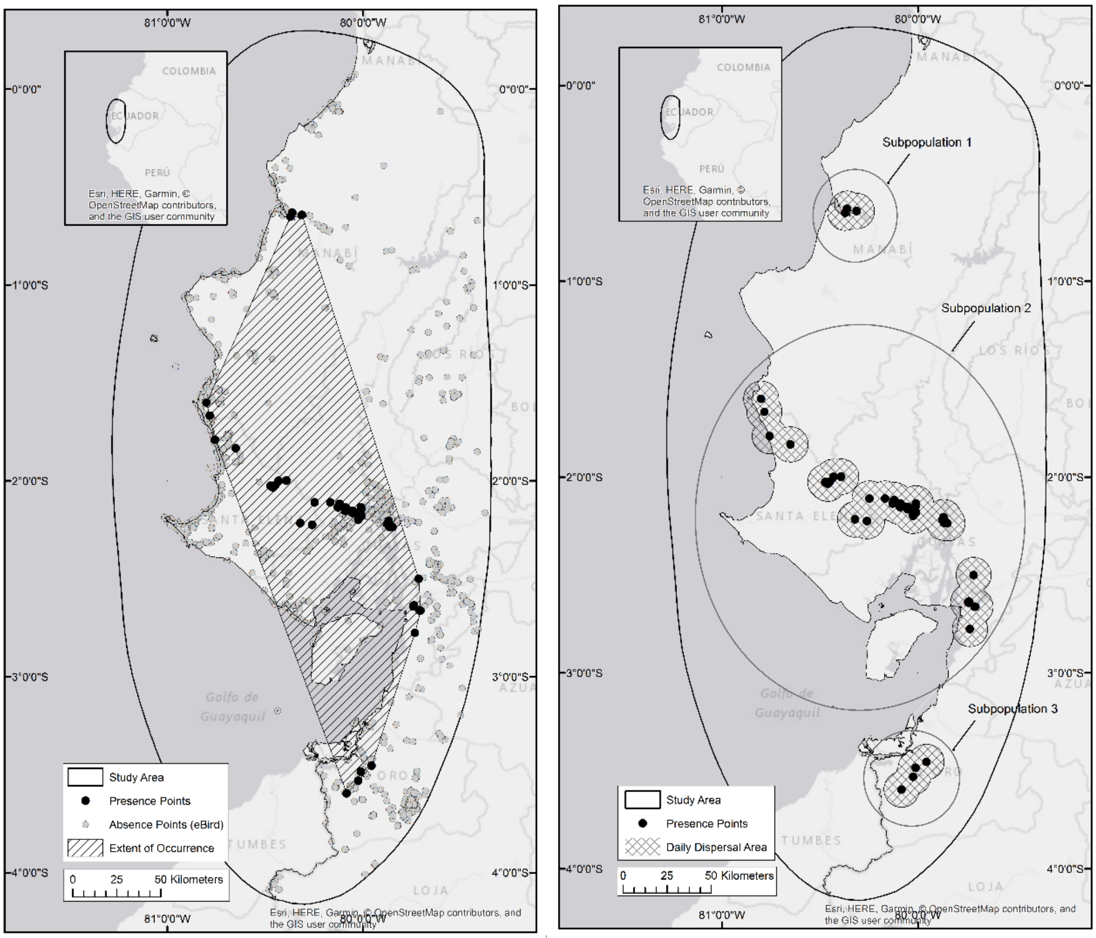


Figure 2a: Presence (n = 132) and absence (n = 4,626) points recorded for *A. lilacina*. Presence points are joined by a convex hull to estimate the species' Extent of Occurrence of 19,890 km². Figure 2b: Occurrence points are surrounded by 10 km buffers to represent a daily dispersal area of 5,313 km², within three subpopulations.

Minimum and maximum counts from each roost survey (Table 1) reflect fluctuations in the number of birds attending each roost during the survey period. Although it is always possible that more roosts exist within the study area, we believe we have identified all remaining large roosts (>30 individuals) and thus we estimate the remaining global population at 741 - 1,090, which includes mature and immature birds. We suggest that counts conducted in March (1,090) at the end of the breeding season, represent the population including young birds, and that counts from January (804) represent the population without breeding birds or at least females with eggs or chicks in the early developmental stages. We saw a slight decrease in global population size between March (1,090) and August (1,046) which may represent juvenile mortality.

Table 1: Number of *A. lilacina* counted during roost surveys at all known roosts. Roost 1 was not present in March 2017. Local reports suggest this was in response to damage caused to the mangrove island roost site by a large earthquake. Thus, the mean of all counts prior to January 2019 is used: 97 and 103 (Nov 2012), 84 and 86 (Aug 2014).

Roost	One	Two	Three	Four	Population estimate
March (2017/18)	93	300 - 480	28 - 110	320 - 407	741 - 1090
Jan (2019)	44 - 95	181 - 338	41 - 72	220 - 299	486 - 804
August (2019)	60 - 116	124 - 144	32 - 57	628 - 729	844 - 1046

When considering all surveys conducted at Roost 3 from 1999–2018, there has been an overall decline in the total number of birds (Figure 3). Our GLMM revealed that average counts declined between the two periods of data collection representing a significant drop in roost size in 2017-2018 compared with 1999-2000 ($\beta = -1.02$, S.E = 0.24, $p < 0.001$). On average, the difference between monthly counts suggests a 60% reduction in the size of the roost. The maximum roost size in 2000 was 229 birds (Berg & Angel 2006), but just 117 in 2018.

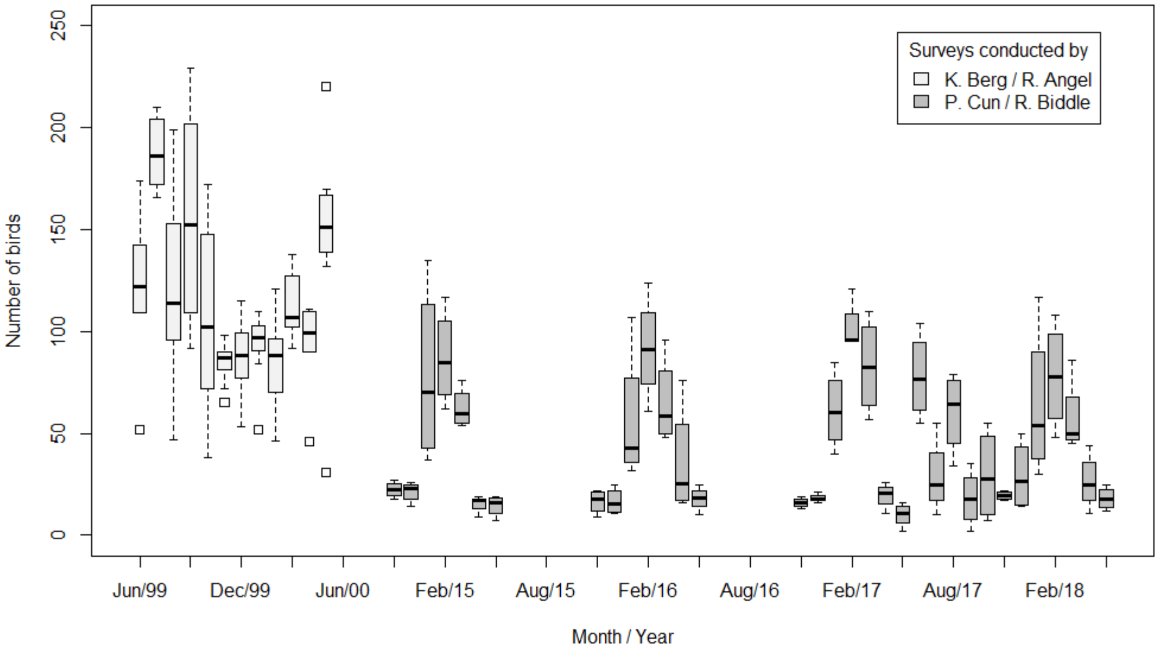


Figure 3: Roost surveys recording the total number of birds departing Roost 3 during morning surveys conducted between June 1999 and May 2018.

Of 52 communities surveyed, presence of pet parrots was confirmed in 37. A total of eleven parrot species, including *A. lilacina* were recorded: Yellow-crowned Amazon *A. ochrocephala,* Orange-winged Amazon *A. amazonica*, Southern Mealy Amazon *A. farinosa*, Red-lored Amazon *A. autumnalis salvini*, Bronze-winged Parrot *Pionus chalcopterus*, Blue-headed Parrot *P. menstruus*, Red-masked Parakeet *Psittacara erythrogenys*, Grey-cheeked Parakeet *Brotogeris pyrrhoptera*, White-winged Parakeet *B. versicolurus* and Pacific Parrotlet *Forpus coelestis*. Communities with pet parrots could be classified (out of bag error rate 16%) using variables of elevation and distance to the nearest roost, it is predicted that 1,617 of the 3,231 additional non-surveyed communities within the study area have a greater than 60% probability of containing pet parrots.

Of the 37 communities with confirmed presence of pet parrots, 17 held pet *A. lilacina.* These 17 could be classified (out of bag error rate 31%) using predictors of distance to roost, distance to sighting and NDVI. It is predicted using this classification, that 79 of the 3,231 additional non-surveyed communities within the study area have a greater than 60% probability of having pet *A. lilacina.* It was felt the value of 60% across the community as a whole, would equate to a much higher probability of at least one pet being owned. From the above, we suggest that within our study area, approximately 1,645 communities have pet parrots, and at least 96 of these have pet *A. lilacina* (Table 2).

Table 2: Of the 52 surveyed communities, 37 reported pet parrots and 17 reported pet *A. lilacina*. Using random forests to predict the occurrence of pet parrots throughout similar communities within the study area, we suggest 1,617 communities have pet parrots and at least 96 have pet *A. lilacina.*

		Pet parrots				Pet *A. lilacina*			
Surveyed communities with confirmed presence		37 / 52				17 / 37			
Out of bag error rate of classification of surveyed communities		15.7%				31.4%			
Classification error table		Confusion matrix:				Confusion matrix:			
			N	Y			N	Y	
		N	10	5	0.33	N	31	4	0.11
		Y	3	33	0.08	Y	12	4	0.75
Mean decrease in accuracy value for each predictor	Community type	2.51				-5.82			
	Distance to sighting	9.45				-1.50			
	Distance to roost	12.95				9.79			
	Distance to road	1.74				-0.88			
	Elevation	16.97				-3.14			
	NDVI	1.98				1.06			
	In protected area or not	4.81				0.17			
Number of communities predicted >60% probability of pets		1,617 / 3,231				79 / 3,231			
Suggested number of communities with pets		1,645				96			

Discussion

We estimate the Extent of Occurrence (EoO) for the recently recognised *Amazona lilacina* to be half of that currently listed on the IUCN Red List, which from available data represents the smallest remaining EoO of any Endangered mainland Amazon parrot (BirdLife International 2018a). We suggest *A. lilacina* has a population size of between 741 and 1,090 birds and that this population is declining, with Roost 3 showing a reduction in size of 60% over the past 19 years – a similar decline to that seen in other parrot species globally (Berkunsky et al. 2017). This rate of decline supports the IUCN listing of Endangered under criterion A, and if reflected over the whole population may qualify the species for listing as Critically Endangered. Further research is needed to assess this, however, when comparing our 2019 counts, to unpublished counts from researchers in 2014, we see a decline of 59% also at Roost 2; an area where strong poaching pressure has been observed (G. Blanco, F. Hiraldo and J.L. Tella, *pers. comm*. 2020). We report that local capture for pets is an ongoing threat, and support the notion that Ecuador should be prioritised for parrot conservation (Olah et al. 2016).

As commonly seen in parrot roost counts, our results showed variability in roost size within and across months. This may be explained in part by imperfect detection whereby birds arrive at or depart roosts undetected due to low light levels or weather conditions (Dénes et al. 2018). Although every attempt was made to account for this, due to the opportunistic nature of some of our roost surveys, a structured counting regime as suggested by Cougill and Marsden (2004) was not always followed. Additionally, it is also possible that some birds may have gathered temporarily in smaller, undetected roosts and thus be missed from main roost surveys. Despite this, we believe the results presented here offer a valuable first estimate of population size and trajectory for this Endangered species.

Amazona lilacina's northern border was previously recorded as southwest Colombia or the Esmeraldas province of Ecuador (CITES 1986; Juniper & Parr 1998; Ridgely & Greenfield 2001a; Forshaw & Knight 2010; Athanas & Greenfield 2016), but we suggest, in agreement with local experts, that these more northern birds are in fact *A. autumnalis salvini* (*pers. comm.*: R. Orrantia, Fundación Jambeli 2013, M. Schaefer, Fundación Jocotoco 2014, R. Ridgely, Rainforest Trust 2015). Within our newly presented EoO we no longer believe the species is restricted to mangrove roosting areas, owing to the discovery of a new roost located >50 km from any mangroves. However, we do suggest that the species is still highly geographically restricted, with an estimated daily dispersal area of just 5,313 km^2 split between three distinct geographically isolated subpopulations. Although movement between these three areas is unlikely due to their separation distance of approximately 40 km, further research into the daily movement and genetic structure of these subpopulations is needed to confirm this.

Historically, threats to this species have been severe: CITES reported thousands of *A. lilacina* being trapped and exported out of the country in the early 1980s (CITES 1986), and Ecuador reported the highest rate of deforestation in South America for the period 2000 – 2005 with the main cause of this being clearing of the lowland coastal forests for agricultural crops (Mosandl et al. 2008). As early as 1986, the plight of this species was highlighted (CITES 1986) and we believe the population is still at risk and in decline. A likely contributor to this is that the range overlaps with a large proportion (46%) of Ecuador's human population (INEC 2010). In addition to the direct threat of local capture for pets, anthropogenic effects such as fire, hunting, land trafficking and the development of squatter settlements are reported as the greatest threats to the lowland coastal forests this species relies on (Horstman 2017).

We predict that over half of all communities within the study area have pet parrots, despite it being illegal since the mid-1980s to hunt or trade species included in the CITES Appendices. Indeed a few of these could be long-lived individuals; however, we expect that to be a minority. Ecuador's confiscation reports also suggest a large number of parrots in captivity with 91% of all birds confiscated between 2003 and 2014 being *Psittacidae* and 7% of these being *A. autumnalis* (Ortiz-von Halle 2018). Law enforcement in the form of pet confiscation does not appear to be a strong deterrent, and only once has a case of bird crime resulted in jail time in Ecuador (Ortiz-von Halle 2018). We predict that nearly 100 communities hold *A. lilacina*, but expect this is an underestimation due to difficulties in identifying parrots to species level. We did gather evidence of poaching of *A. lilacina* chicks and adults during fieldwork, either to generate core income, or incidentally, to fulfil a specific economic need such as buying uniforms at the start of the school term. Additionally, reports of farmers using nets or poison to protect their crops against parrots and historical reports of family relatives shooting macaws and Amazons for food were made. Although legal subsistence hunting does not appear a threat to the species, recent concern has been raised regarding its sustainability in the light of changes in human population size, hunting methods, and habitat fragmentation (Suarez & Ríos 2019).

The lowland coastal provinces where *A. lilacina* occurs have been identified as having an acute lack of protected areas (Cuesta et al. 2017). Additionally, the lowland forests, mangroves and algarrobo trees, are all habitats essential for local community income and sustenance be it through hunting, fishing for crabs, cutting of firewood or making charcoal. Outside of these habitats, our observations occasionally recorded the species in crop fields, gardens, and even villages where fruit trees have been planted, and in the last couple of years eBird users are more frequently recording the species within the large city of Guayaquil. This species is clearly existing across a highly anthropogenically influenced landscape, and although there are examples worldwide of parrots adapting to such environments (Lill 2009; Martens & Woog 2017) the effects this may have on their natural behaviours or ecological functions could be significant (Luna et al. 2018).

Conclusion and further research

Following the analysis of the data presented in this chapter and published by Biddle et al 2020, a reassessment was made of *Amazona lilacina's* Red List status. The species no longer met the criteria of Endangered based on the reported population size and that of the subpopulations (IUCN 2012). The Extent of Occurrence and population size now met the threshold for Vulnerable, whilst the population trajectory met the criteria of Critically Endangered (IUCN 2012). The 60% decline over 20 years that we reported in one subpopulation was likely to equate to 82% decline over three generations (37 years). After an open online discussion forum and further evidence suggesting that this decline was consistent across the species range, *A. lilacina* was up-listed to Critically Endangered (BirdLife International 2020b).

Another key finding from this chapter was that *A. lilacina* did not only occur in areas nearby to mangroves where they could communally roost. This meant that there could be, previously overlooked, suitable inland habitat. Therefore, the next research priority was to assess the species distribution. Numerous studies have shown that local ecological knowledge can provide extremely useful information about species distributions, particularly for species that are cryptic and hard to observe (Melovski et al. 2018; Díaz-Ruiz et al. 2019; Ghoshal et al. 2019; Skroblin et al 2019). We believed this to be especially likely for *A. lilacina*, due to their gregarious and noisy roosting behaviour. Despite it being extremely hard to observe foraging or nesting birds, anyone living nearby to a roost is likely to hear them arrive and depart each day (Ridgely & Greenfield 2001a; Enkerlin-Hoeflich et al. 2006). Therefore, researcher–led questionnaires were conducted, across the range to gather people's observations of parrots in their local area. The results of these and field observations from within the species extent of occurrence were then analysed and used to build species distribution models, to predict areas of suitable habitat – see Chapter Three.

Chapter Three: The value of local community knowledge in species distribution modelling for *Amazona lilacina*.

Biddle R, Solis-Ponce I, Jones M, Marsden S, Pilgrim M, Devenish C. 2021. The value of local community knowledge in species distribution modelling for a threatened Neotropical parrot. Biodiversity and Conservation **30**:1803-1823.

Abstract

Species distribution models are widely used in conservation planning, but obtaining the necessary occurrence data can be challenging, particularly for rare species. In these cases, citizen science may provide insight into species distributions. To understand further the distribution of the newly described and Critically Endangered *Amazona lilacina,* we collated species observations and reliable eBird records from 2010 – 2020. We combined these with environmental predictors and either randomly generated background points or absence points generated from eBird checklists, to build distribution models using MaxEnt. We also conducted interviews with people local to the species' range to gather community-sourced occurrence data. We grouped these data according to perceived expertise of the observer, based on the ability to identify *A. lilacina* and its distinguishing features, knowledge of its ecology, overall awareness of parrot biodiversity, and the observation type. We evaluated all models using AUC and Tjur R^2. Field data models built using background points performed better than those using eBird absence points (AUC=0.80+/-0.02, Tjur R^2=0.46+/- 0.01 compared to AUC=0.78+/-0.03, Tjur R^2=0.43+/- 0.21). The best performing community data model used presence records from people who were able recognise a photograph of *A. lilacina* and correctly describe its distinguishing physical or behavioural characteristics (AUC=0.84+/- 0.05, Tjur R^2=0.51+/- 0.01). There was up to 92% overlap between the field data and community data models, which when combined, predicted 17,772 km² of suitable habitat. Use of community knowledge offers a cost-efficient method to obtain data for species distribution modelling; we offer recommendations on how to assess its performance and present a final map of potential distribution for *A. lilacina.*

Introduction

Understanding species distributions is essential for conservation planning (Wilson et al. 2005) but for species that are rare, sparsely distributed, or inconspicuous, this information is often lacking. In such cases, species distribution models (SDMs) and their outputs, can be particularly useful, as long as they are based on ecological theory and built using accurate data (Guisan & Thuiller 2005). SDMs allow the probability of occurrence to be predicted in un-surveyed areas, which can inform future field investigations and have many important conservation applications (e.g. Pearce & Lindenmayer 1998; Araújo et al. 2004). For all SDMs, species presence data are needed. Traditionally this comes from direct species observations or museum records, but more recently scientists have looked to integrate different sources of data, such as citizen science, to make better inferences of the true distribution of species (Amano et al. 2016; Coxen et al. 2017; Fletcher et al. 2019; Steen et al. 2019; Isaac et al. 2020).

The quality of outputs gained from SDMs is affected by factors such as data type, sampling bias and imperfect detection (Lahoz-Monfort et al. 2014; Guillera-Arroita et al. 2015). MaxEnt is one of the most commonly used methods for deriving SDMs and has been shown to produce useful models even when dealing with small sample sizes (Wisz *et al*., 2008; Elia et al. 2015). Whilst other methods require absence data to be collected, MaxEnt uses presence data combined with a background sample drawn randomly from the study area (Phillips et al. 2006, Phillips & Dudík, 2008; Elith et al., 2011). Both presence-absence and presence-background data methods have limitations; namely that presence data often do not represent an unbiased sample of locations at which the species is present, and that absence data can lead to the inclusion of false absences (Guillera-Arroita et al. 2015). These limitations must be considered against the proposed use of model outputs; for instance, presence-background data may be sufficient when outputs are to be used to direct further field investigations, but insufficient if outputs are to directly inform land management for conservation (Lahoz-Monfort et al. 2014). The predictive ability of models may also be reduced if imperfect detection is not accounted for, and may result in outputs being more likely to predict areas in which the species is easier to observe, rather than where it is more likely to occur. It is therefore essential that the effects of imperfect detection are minimised by ensuring a sufficiently large sampling effort at surveyed locations (Lahoz-Monfort et al. 2014)

For species where field observations are lacking, citizen science data is a valuable and widely used resource (Brook & McLachlan 2008) which can help determine species presence, absence or abundance (Melovski et al. 2018; Díaz-Ruiz et al. 2019; Ghoshal et al. 2019; Skroblin et al 2019). Some methods allow large volumes of data to be collected more cost effectively than traditional field survey methods, for example postal surveys (FitzGibbon & Jones 2006), telephone interviews (Mallory et al. 2003) and social media (Pace et al. 2019). Often this data is used to supplement ‘expert’ data by guiding further field surveys (Hart & Upoki 1997; O’Brien et al 1998; Chaiyes et al 2017) but in some cases it is shown to be just as accurate as the equivalent ‘expert’ data, providing that some form of filter for reliability is incorporated (Polfus et al. 2014). Recently, a number of studies have even shown that georeferenced occurrence data collected through citizen science platforms and online biodiversity databases such as eBird, can be used to build accurate SDMs (Bradsworth et al. 2017; Coxen et al. 2017; Fournier et al. 2017; Saunders et al. 2020). However, it is important to note that all opportunistically collected citizen science data present additional challenges such as spatial biases and variation in observer skill (Isaac & Pocock 2015; Johnston et al. 2020) and online recording schemes such as eBird create barriers by requiring observations to be collected and submitted in a particular way.

Within all types of citizen science data, there is variation in accuracy. For example studies have shown that ‘freelisting’ (Bernard 2006), a quick survey method where participants are asked to list the species they see in their local area, can result in people reporting species that do not occur and omitting ones that do (Can & Togan 2009; Díaz-Ruiz et al. 2019). However, the cost efficiency of citizen science may compensate for reduced accuracy depending the data collected and extent of errors (Gardiner et al. 2012). If citizen science data are to be used to infer information about distribution, and as input data for the creation of SDMs, some method of boosting data accuracy or accounting for level of expertise is essential (Kosmala et al. 2016; Johnston et al. 2019). Previous studies have used prior selection of participants i.e. only interviewing key informants selected by community leaders due to their perceived expertise (Mallory et al. 2003; Lopes et al. 2018). Others have developed some kind of scoring system, to determine data accuracy (Frey et al. 2013) by only regarding contributions from participants who are able to recognise photographs of the study species and provide accurate location information (Ghoshal et al. 2019), or by using photographs of non-native species to assess participants identification skills (O’Brien et al. 1998).

To further our understanding of the distribution of a newly described and Critically Endangered parrot species *Amazona lilacina* (Biddle et al. 2020; BirdLife International 2020b), we:

i. collate all known species locality records, including from our own observations, reports from expert ornithologists, and reliable eBird records (2010-2020) and use these to build distribution models;

ii. collect data on local peoples' experiences and observations of wild *A. lilacina* through structured face-to-face interviews;

iii. assess community survey data based on different quality filters and use these data to build distribution models;

iv. determine the best performing distribution models built from species records and community reports, and compare their outputs in order to direct future field investigation.

Methods

Study area

Amazona lilacina, a species recently split from the *A. autumnalis* group, is found in the coastal region of Ecuador where its small population is sparsely distributed around dry forests and mangrove ecosystems (Biddle et al 2020). These habitats are described as amongst the most imperilled ecosystems on earth (Dodson & Gentry 1991). During the day time *A. lilacina* is highly inconspicuous, feeding silently in the forest canopy in small groups which presents difficulty in using traditional field surveys methods to collect presence data (Ridgely & Greenfield 2001a). However, in the evenings birds will form conspicuous groups and fly to communal roost sites (Berg &

Angel 2006) which means that communities anywhere on this flight path, are often aware of the species presence.

The rural coastal communities are considered to be in the most deprived areas of Ecuador, with almost one quarter of all people living in multidimensional poverty (Mideros 2012). The deprivation gap regarding food and water, education, communication, and housing, is greater here than in any other part of the country (Mideros 2012). Within our sampled communities, (Figure 1a) people mainly make a living as farmers, fishers or crab fishers, and 60% have either none, or only primary level schooling. Many communities in this region are highly inaccessible, especially in the rainy season and 57% of people we surveyed had lived in their village their entire lives. The flow of information into and out of these communities is reported to be almost non-existent, with only 40% of households having access to one form of telecommunication (radio, television, phone, computer) (Mideros, 2012).

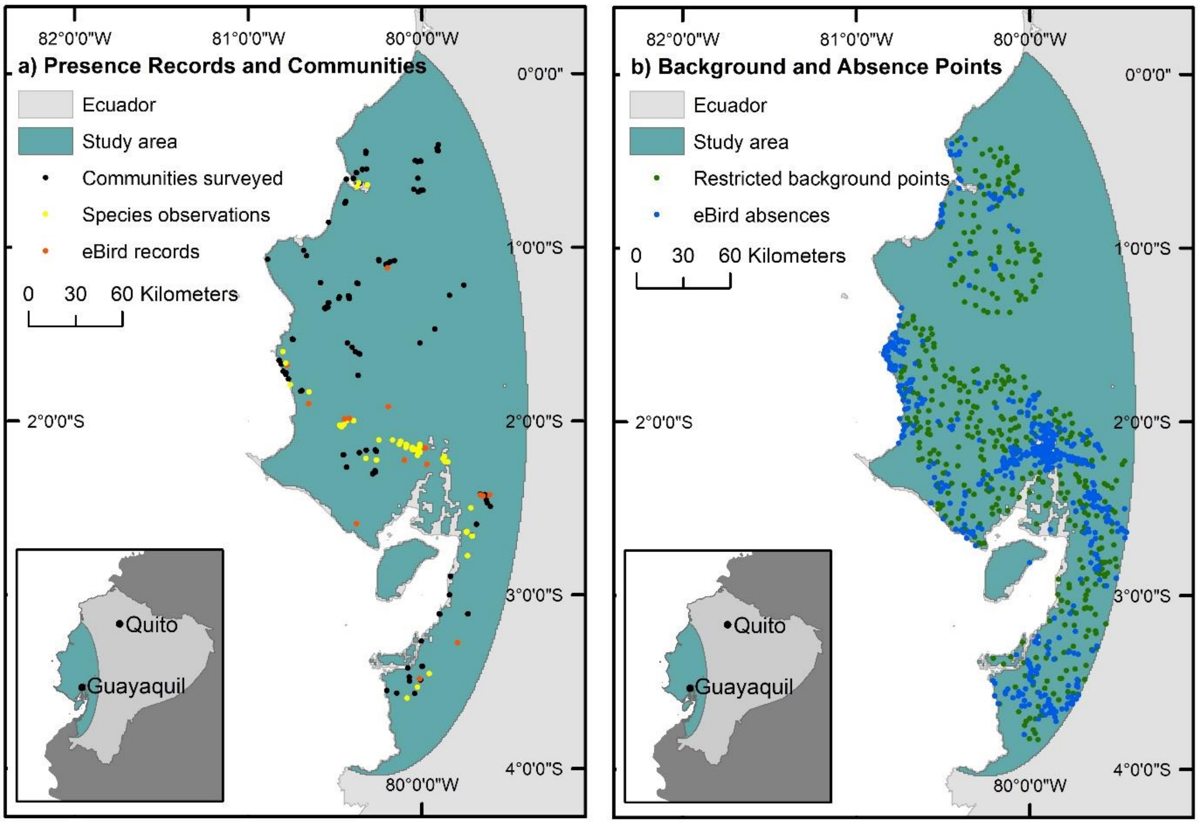


Figure 1 (1a): Locations of all households taking part in interviews, all records of *Amazona lilacina* collated between 2010 – 2020 and, (1b): eBird absence points, representing all complete checklists that did not report *A. lilacina*, and random background points matching the number of eBird absence points available, within a 30km buffer of all *A. lilacina* presence records.

Field observations and eBird records

Observational data were collected during ten field trips led by RB, lasting two to three weeks each (November 2012, January and August 2014, November 2015, August 2016, January and March 2017, February 2018, January and August 2019). Data collection was informed by: 1) existing information on known distribution and habitat use (Juniper & Parr 1998; Ridgely & Greenfield 2001; Berg & Angel 2006; Forshaw & Knight 2010; Athanas & Greenfield 2016); 2) information on habitat distribution from Google Earth and the Ministerio del Ambiente ecosystem map; 3) direct communication with local NGOs, ornithologists, local guides and bird tour companies. All sightings of perched *A. lilacina* made by RB, ISP, MP, Fundación Pro-Bosque staff, Fundación Jambeli staff, and Juan Freile between 2010 and 2020 were georeferenced (sightings of birds in flight were omitted).

All eBird data for Ecuador, including observations and sampling data were downloaded in December 2020. To ensure that no records were missed due to changing taxonomic nomenclature, data were filtered to include all birds recorded as *A. autumnalis* (which included *A. a. lilacina* and *A. a. salvini)* between 01/01/2010 and 31/12/2020. Records that were not deemed as *A. lilacina* based on either photographic evidence or location (i.e. within the Esmeraldas province) were removed, as were records that were already represented by our own observations (within 1 km). To avoid misrepresentation of location, all records that were reported as “general area” which implies the record does not correspond to that exact location were removed, as were records with survey effort >5 hours and >5 km in length (Johnston et al 2019). Finally, locations of parrots within urban locations in the big city of Guayaquil (visualised on Google Earth) were removed to avoid escaped pets or captive birds being included in models.

Distribution models from field observations and eBird records

The MaxEnt function of the package 'dismo' (Hijmans et al. 2020) in R (version 3.6.0, R Core Team, 2019) was used to create species distribution models from field observations and eBird records, referred to from now on as the field models. These were first built using eBird absence points generated by filtering for all complete checklists within our study area that did not report the presence of *A. autumnalis* (*A. a. salvini* or *A. a. lilacina*) (Figure 1b). Absence points were also limited to checklists that were <5 km in length, <5 hours in duration and with fewer than ten observers (Johnston et al. 2019), and to a buffer of 30 km from all field observations and eBird records. Our second and third field models were built using random background points generated in ArcGIS (Version 10.8.1) from within the same buffer: the second model had 4,597 and the third had the same number as eBird absences available (458). Spatial autocorrelation was controlled for by limiting points to one per 1 km using the R package 'spThin' (Aiello-Lammens et al. 2015). A set of interpolated bioclimatic predictor variables available from WorldClim (https://www.worldclim.com/bioclim) representing different measures of temperature and rainfall, plus additional predictors thought to have some biological significance for the species were used: Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product over the period 2010-2015 as a proxy of vegetation cover; distance to mangrove (Hamilton & Casey 2016) and distance to the nearest river (Military Geographic Institute, IGM). Predictors were checked for pairwise correlation across random points within the study area, using pair plots (Zuur et al. 2010); where correlation coefficients between pairs of predictors were >=0.70, the less biologically meaningful predictor was removed. The final variables were; distance to the mangrove, distance to a river, annual mean NDVI and NDVI seasonality, mean diurnal temperature range, annual mean temperature and temperature seasonality, precipitation of wettest month, precipitation of coldest quarter and precipitation of driest month. To allow comparison between the field and community models, we averaged predictor values across 9 km² at all points used in all models to reflect respondents' reference to their 'local area', which could encompass areas of community owned land >1 km away from their house. To ensure this did not affect model outputs or accuracy we trialled models built using predictor values at the exact location, compared to those averaged over 9 km², and found no difference.

Models were evaluated with AUC and Tjur R² (Tjur 2009) over five-fold cross validation; the mean evaluation metrics and their standard deviation are presented. AUC measures how well model predictions discriminate between presence and absence (Wisz et al. 2008). Tjur R² represents the difference between the mean model value at the presence locations and the mean value at the absence / background locations. All the data were included in the final models. Finally, we present variable importance scores, with permutation values >10%, with a high value indicating that the model depends heavily on that variable (Phillips et al. 2006) and response plots for the most accurate field model.

Community questionnaires and response filtering

Researcher–led questionnaires were carried out to identify areas that were reported by local people to be occupied by *A. lilacina*. Communities were chosen to be included in this study due to their close proximity to dry lowland forests (within approximately 10 km), identified using the Ministerio del Ambiente ecosystem map. Furthermore, all communities surveyed were inside or within 70 km of the species Extent of Occurrence (Biddle et al 2020). A pilot study was conducted after which interviews were carried out in January-July 2017. Questionnaires were conducted in Spanish by a local Ecuadorian researcher (ISP), with only the interviewer and respondent present (Tourangeau & Yan 2007). We aimed to survey a minimum of three households per community representing a cross section of demographic groups, but often this depended on the availability of participants and the size of the community. In all cases, prior verbal consent was obtained, and although less than fifteen people did not complete interviews, interviewees could decline from contributing once the purpose of the research was explained (Appendix 1).

The location of each questionnaire, normally by the participant's house, was recorded and participants were asked to respond with reference to their immediate local area which included their house, garden, and local community land. Demographic information regarding age, gender, level of schooling, and how long they had lived in the village, was collected, but interviews were anonymous, and data were coded to ensure that no individuals could be identified. Interviewees were not made aware of the species in concern before starting the interview, during which they were asked to name and describe which parrot species (if any) they see in their local area, then confirm from a selection of ten parrot photographs (the order of which was rotated at random between surveys) (Table 1). If a participant confirmed they currently (within the last year) see *A. lilacina* at their location, they were then asked a number of questions designed to help assess the accuracy of this information. Each interview (Appendix 2) took approximately 20 minutes to complete.

Table 1: Photographs of ten parrot species were presented to questionnaire participants. The chance of observation is rated as likely (if the species range covers the entire study area), possible (if the species range covers more than half of the study area) or unlikely (if the species range covers less than half of the study area) (Freile & Restall 2018).

Parrot species	Chance of observation	Species status
1. Southern Mealy Amazon *Amazona farinosa*	Unlikely	Rare
2. Blue-headed Parrot *Pionus menstruus*	Possible	Common
3. Bronze-winged Parrot *Pionus chalcopterus*	Possible	Uncommon
4. Red-masked Parakeet *Psittacara erythrogenys*	Likely	Common
5. Great Green Macaw *Ara ambiguous*	Unlikely	Very rare
6. Pacific Parrotlet *Forpus coelestis*	Likely	Common
7. Grey-cheeked Parakeet *Brotogeris pyrrhoptera*	Possible	Uncommon
8. Lilacine or Ecuadorian Amazon *Amazona lilacina*	Possible	Rare
9. Brown-headed Parrot Poicephalus cryptoxanthus	Not present	Not present
10. Yellow-crowned Amazon *Amazona ochrocephala*	Not present	Not present

To examine the influence of accuracy of community data, we filtered responses according to the ability to recognise the species, knowledge of its distinguishing features, overall awareness of parrot biodiversity, and observation type (i.e., if the bird was seen flying, nesting, perched or feeding). We created six groups of responses to represent realistic scenarios that may be used to select which observations to include in distribution investigations (Table 2). We created a further eleven groups which represented all possible combinations of groups three to six, for example group seven represented a group of participants who had answered correctly for all of groups three, four, five and six (Appendix 3).

Table 2: Six methods of community data filtering were developed to group community responses. All possible combinations of response groups three – six were included, resulting in 17 groups of community data from which to build species distribution models.

Community data group	Question	Answer suggesting accurate presence report
1.	Can you name the different parrot species you see in your local area?	Gives any name used locally for *A. lilacina.*
2.	Can you confirm from the following photographs, which parrot species occur in your local area?	Recognises and identifies the *A. lilacina* photograph.
3.	Describe how you distinguish *A. lilacina* from other parrot species around here.	Confirms *A. lilacina* is present from photograph or naming, and also provides one or more physical (description of the overall body shape i.e. square tail, medium size, red forehead, black beak) or behavioural (migration behaviour and associated distinct vocalisation) characteristic specific to *A. lilacina.*
4.	What habitat do you think *A. lilacina* prefers?	Confirms *A. lilacina* is present from photograph or naming, and also describes preferred habitat as forest or mangrove ecosystem.
5.	Can you confirm from the following photographs, which parrot species occur in your local area?	Confirms *A. lilacina* is present from photograph or naming, and also recognises the Red-masked Parakeet *Psittacara erythrogenys* and the Pacific Parrotlet *Forpus coelestis* (which are both common and likely to be observed).
6.	What were the *A. lilacina* doing in your local area? (flying / nesting / feeding / perched / something else).	Confirms *A. lilacina* is present from photograph or naming, and also confirms it was either perched, nesting or feeding (i.e. not just flying over).

Distribution models from community data

We created distribution models based on groups of community data with varying levels of accuracy as listed in Table 2; the community models. Each participant's response was associated with a location representing a 1 km² pixel on our distribution maps. These presence locations were combined with environmental variables and background points following the same methods as for the field model. All background points were restricted to buffers of 30 km from community survey presence points. We averaged predictor values across the 9 km^2, as for the field model, to reflect respondents' reference to their 'local area', which could encompass additional areas of community owned land. In order to evaluate the accuracy of the community data models, we use the same methods as for the field models; AUC and Tjur R^2 (Tjur 2009) over five-fold cross validation. We present these, alongside permutation values where their contribution to the model is >10% for all models, and the habitat suitability output and response plots for the best performing model.

Model comparison

Once we had identified the best performing field observation model and community data models, we compared the overlap between their habitat suitability outputs. These outputs are interpreted as maps of potential distribution with values indicating the level of habitat suitability for each pixel, on a scale of zero to one. There are several methods used to compare model outputs (Galante et al. 2018). We chose Moran's I which represents the difference between suitability values at each cell, and the relative rank coefficient which estimates the probability that the relative suitability ranking for a patch of habitat cells is the same for the two models (Warren et al. 2008; Warren & Seifert 2011). We calculated these using the niche overlap function in ENMTools (Warren et al. 2010). Both methods produce metrics which range from zero (no overlap) to one (complete overlap).

To predict areas of potential distribution, it was necessary to classify areas as either 'suitable' or 'unsuitable' depending on their model value. Many thresholding rules are justified for presence-only occurrence data (Peterson et al. 2011). We chose the 10%

omission rate threshold (Galante et al. 2018) where the model value which includes 90% of the values predicted at the presence locations used to create that model, is applied as a threshold to the habitat suitability output to distinguish between presence and absence. We calculated and applied this independently to the two best performing models. We present a final combined map of distribution that represents areas predicted as suitable or not by either of the final models. We extracted the values for the top three predictor variables from the best performing models, in areas where both models predicted presence, compared to areas where only the field model or only the community model did, and plotted these using the R package ‘ggplot2’ (Wickham 2016).

Predictors of community data performance

Once the best performing community data model been determined, a generalized linear mixed model (GLMM) was conducted in R (version 3.6.0, R Core Team, 2019) using the package ‘lme4’ (Bates et al. 2020). The binomial response of whether or not a participant was included in the response group used to build that model was analysed to determine any effects of participants’ social demographics: gender, level of schooling, age and number of years in the village. Only communities where at least one wild *A. lilacina* observation had been reported were included, and the community location was included as a random effect. We checked for correlation between the age and number of years spent in the village using Pearson's product-moment correlation, and between gender and level of schooling (some or none) using a Chi-squared test of independence, and only included non-correlated variables in our GLMM.

Results

Field observations and distribution model

Our field observations generated a total of 132 occurrence points. A further 14 locations from eBird were included, to create a final dataset of 146 *A. lilacina* presence locations. These were reduced to 59 (47 field observations and 12 eBird records) during the spatial rarefication process, combined with either: 458 eBird absence points (model one); 4,597 randomly generated background points (model two) or; 458 randomly generated background points (model three) and entered into model building with the ten non-correlated predictor variables. The resulting mean of five-fold cross validation AUCs were 0.78+/-0.03, 0.80+/-0.02, 0.79+/-0.02 and the resulting mean of five-fold cross validation Tjur R^2s were 0.43+/-0.21, 0.46+/-0.01 and 0.41+/-0.01 for models one to three, respectively. Therefore, field model two was considered to be the best performing model (Table 3). The habitat suitability output from model two shows that the suitable habitat follows the Chongón Colonche mountains range, from Guayaquil north-west towards the coast, with additional suitable areas in the far south of the country bordering Peru, and the north of the study area in mid-Manabí (Figure 2a). Environmental variables that showed a permutation importance of >10% were annual mean NDVI, distance to the mangrove, and temperature seasonality and response plots (Figure 2b) suggest that suitability of habitat is associated with close distance to mangrove and a relatively high annual mean NDVI.

Table 3: Results of the three field models showing permutation values of environmental where >10%. Based on AUC and Tjur R^2 values, model two (in bold) is the best performing field data model.

Model	Sample size (after thinning)	Permutation importance of variables >10%	Mean of five-fold cross validation AUC (+/-sd)	Mean of five-fold cross validation Tjur R² (+/-sd)
Field model 1 with eBird absences (458)	146 (59)	Mean annual NDVI (34%) Precipitation of coldest quarter (19%) Precipitation of wettest month (16%) Mean diurnal temperature range (14%)	0.78 (+/-0.03)	0.43 (+/- 0.21)
Field model 2 with background points (4,597)	**146 (59)**	**Mean annual NDVI (32%)** **Distance to mangrove (24%)** **Temperature seasonality (17%)**	**0.80 (+/-0.02)**	**0.46 (+/- 0.01)**
Field model 3 with restricted background points (458)	146 (59)	Mean annual NDVI (25%) Distance to mangrove (24%) Temperature seasonality (21%) Precipitation of coldest quarter (20%)	0.79 (+/-0.02)	0.41 (+/- 0.01)

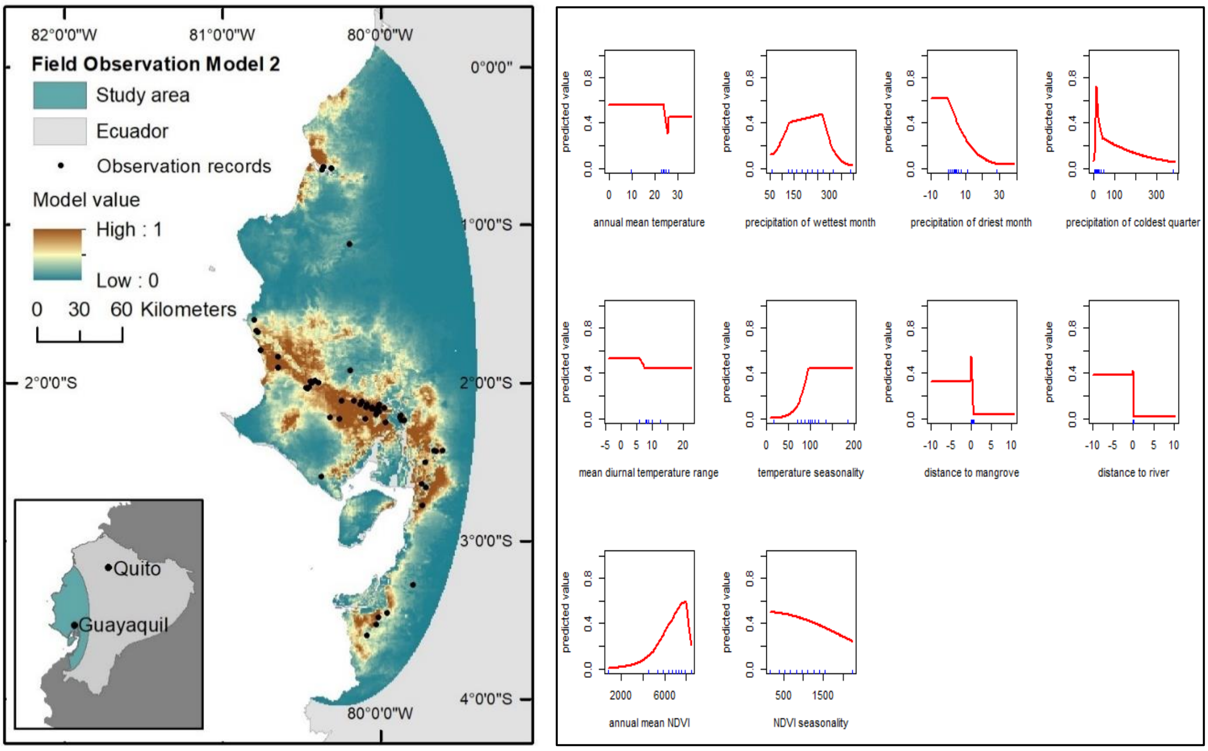


Figure 2 (2a): The habitat suitability output from the best performing field model which is built using 59 species records and 4,597 background points (2b): The variable response plots for this model.

Community questionnaires and reliability scoring

A total of 404 people from 72 communities took part in questionnaires, including 183 women and 221 men, with an average of 5.6 households per community (min two, max 23). There was a variety of schooling levels, from none (31), primary (214), secondary (128), to university (31) and in how long participants had lived in their community (1-84 years) but the majority (88%) had lived there for ten or more years. Of the 404 participants, 393 reported seeing parrots in general. Although it was posed in our questionnaires that participants should answer with reference to birds seen in the wild, when asked “where did you see this bird?” 15 respondents replied “as a pet” - these 15 responses were removed from the community models.

Distribution models from community data

After filtering community data based on the six groups in Table 1, and creating combination groups where participants answered positively for multiple categories, each group had a sample size of >=27 (27–155). After spatial thinning all datasets contained >=18 (18-67) georeferenced occurrence points. Each group of points was combined with 3,931 background points and the same ten non-correlated predictor variables as those included in the field model. Models were built based on groups one to six of data, and then all eleven possible combinations of groups three to six. None of the combination models improved the performance of the model (Appendix 3). The mean of five-fold cross validation AUC for the six main models was >0.74+/-0.03 and Tjur R^2 >0.39+/-0.02. Based on these values, model three is the best performing community model (Table 4). The habitat suitability map of community model three shows a similar area of suitable habitat to the field data model, but with additional increased suitability predicted along the coastline (Figure 3a). Environmental variables with a permutation importance of >10% were distance to the mangrove and temperature seasonality, and response plots for this model suggest that suitability of habitat is associated with areas closer to mangroves (Figure 3b).

Table 4: Georeferenced occurrence points for each group of community data were thinned to control for spatial correlation. The resulting models all showed high accuracy (AUC >0.74+/-0.03). Based on the AUC and Tjur R^2, model three (in bold) is the best performing community data model.

Model	Sample size (after thinning)	Permutation importance of variables >10%	Mean of five-fold cross validation AUC (+/- sd)	Mean of five-fold cross validation Tjur R^2 (+/- sd)
Community model 1	27 (18)	Distance to mangrove (52%) Distance to river (14%) NDVI seasonality (20%)	0.83 (+/- 0.11)	0.48 (+/- 0.06)
Community model 2	155 (67)	Distance to mangrove (35%) Precipitation of wettest month (17%)	0.77 (+/- 0.10)	0.42 (+/- 0.03)
Community model 3	**115 (53)**	**Distance to mangrove (54%)** **Temperature seasonality (12%)**	**0.84 (+/- 0.05)**	**0.51** (+/- **0.01)**
Community model 4	134 (64)	Distance to mangrove (43%) Precipitation of wettest month (22%)	0.76 (+/- 0.06)	0.41 (+/- 0.01)
Community model 5	67 (40)	Distance to mangrove (53%)	0.76 (+/- 0.07)	0.40 (+/- 0.03)
Community model 6	40 (26)	Distance to mangrove (37%) Precipitation of wettest month (19%)	0.74 (+/- 0.03)	0.39 (+/- 0.02)

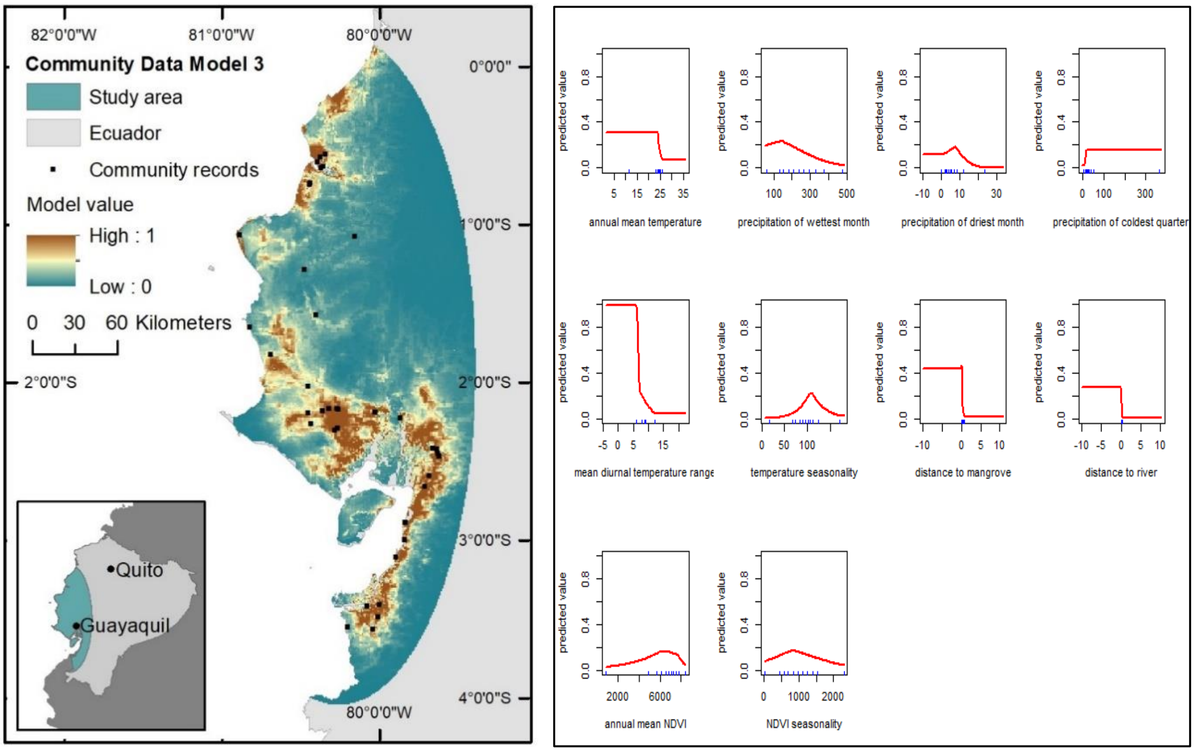


Figure 3 (3a): The habitat suitability output from the best performing community data model, built using 53 reports where participants were able to recognise a photograph of the species and provide one or more physical or behavioural characteristic specific to *A. lilacina.* (3b): The variable response plots for this model.

Model comparison

After calculating and applying thresholds to the best performing field and community models, the field model predicts 13,969 km² of suitable habitat and the community model predicts 13,067 km² (Table 5). When we combine these threshold habitat suitability outputs, they overlap in 9,314 km² of predicted suitable habitat, the community data model predicts a further 3,753 km² that the field data does not, and the field data model predicts a further 4,655 km² that the community model does not (Figure 4). The top three predictor variables from both of these models were; distance to mangrove, temperature seasonality and mean annual NDVI. When plotting the values from predicted presence areas by both models, just the field model or just the community model, areas that are predicted by only the community model only have a slightly lower mean annual NDVI and are closer to mangroves (Figure 5). There is a high level of overlap between the field data and community data habitat suitability outputs (before applying a threshold). The relative rank coefficient, which estimates the probability that the relative suitability ranking for a patch of habitat cells is the same for the two models, is 0.82, and the Moran's I, which represents the difference between suitability values at each cell, is 0.92 (Table 5).

Table 5: The area of predicted presence by the best performing field and community data models after calculating and applying a 10% omission threshold and the level of overlap between the two habitat suitability outputs before applying a threshold.

Most accurate models	10% omission threshold value	Predicted presence area (km²)	Level of overlap: relative rank	Level of overlap: Moran's I
Field data model 2	0.30	13,969	0.82	0.92
Community data model 3	0.46	13,067		

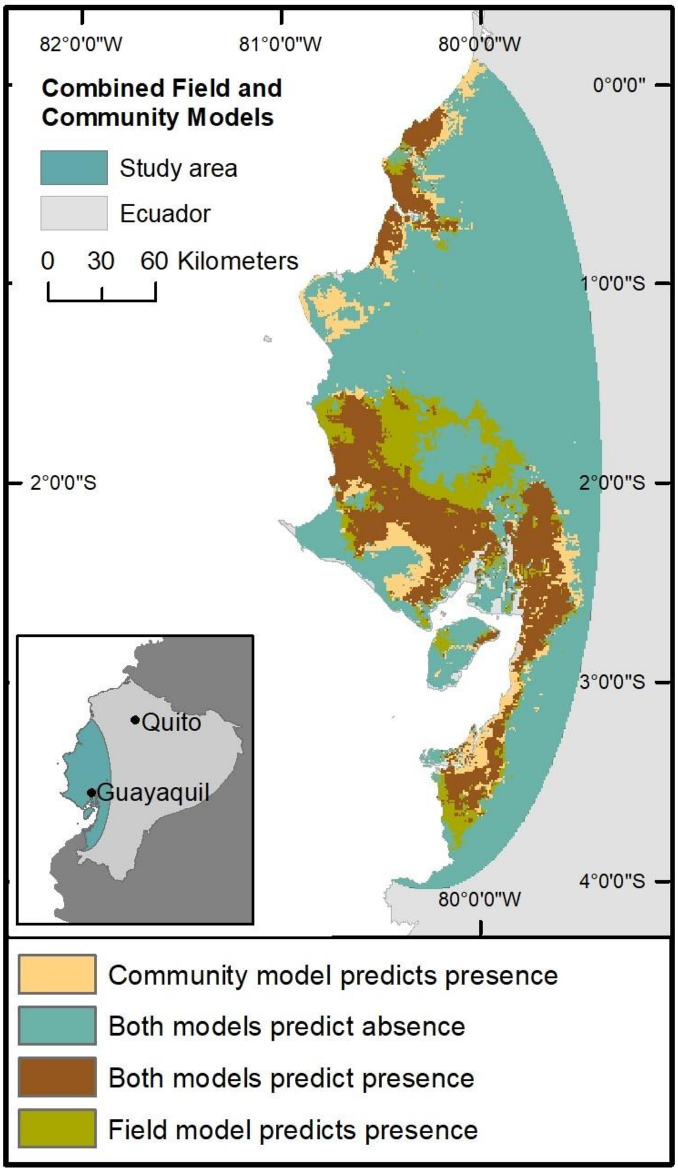


Figure 4: After calculating and applying thresholds independently to the two best performing models, their predicted suitable habitat overlaps in 9,314 km², but the community data model predicts a further 3,753 km² that is suitable, and the field data model predicts a further 4,655 km² that is suitable for *A. lilacina*.

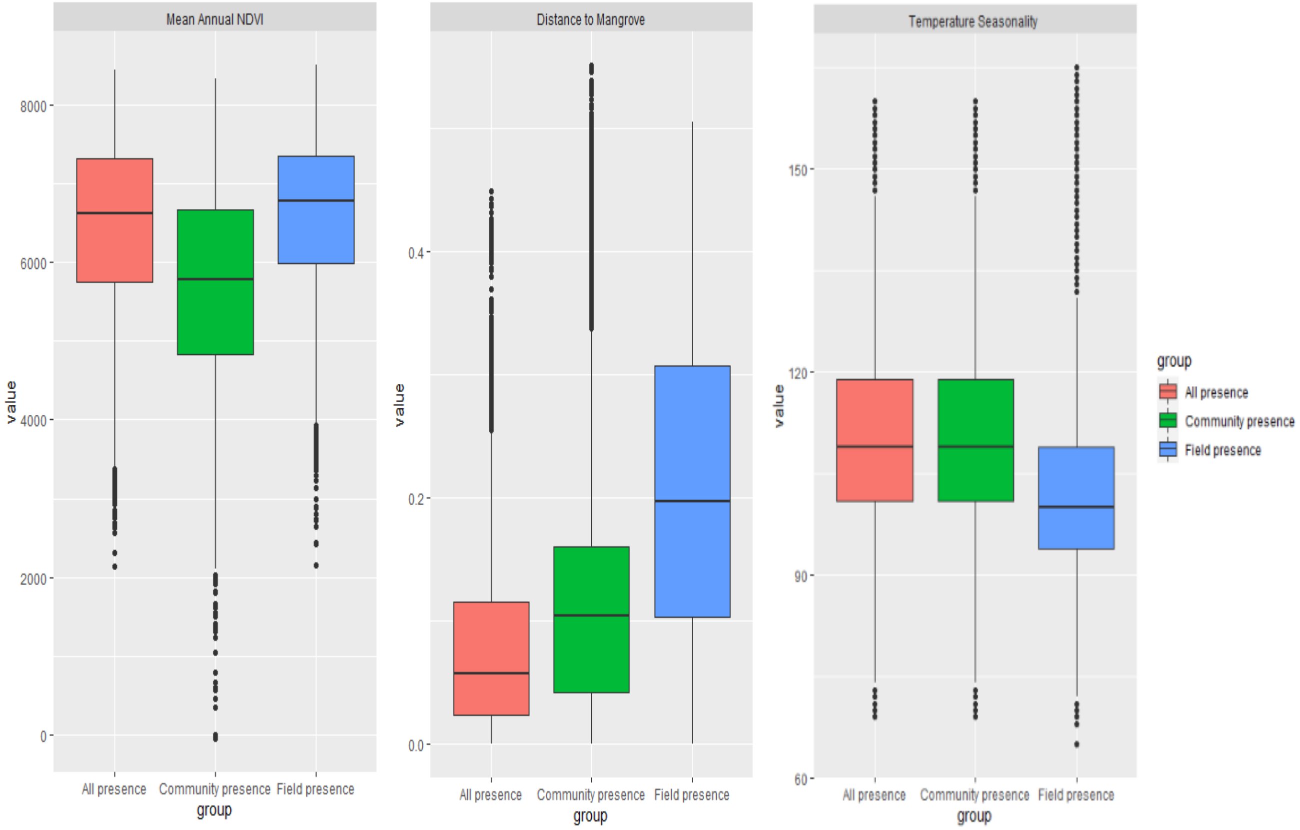


Figure 5: Box plots showing predictor values in areas predicted as suitable (after applying a threshold) by both the best performing community and field data models, only the field data model, and only the community data model. The predictors with a permutation importance of >10% in the final models were included; mean annual NDVI (5a) Distance to mangrove (5b) and temperature seasonality (5c).

Predictors of community data performance

Of the 52 communities where at least one observation of wild *A. lilacina* was made, and thus species presence was likely, 35% (105/304) of participants were included in community data group with the best model performance. These 105 participants (70 men and 35 women) were able to either name or recognise a photo of the species, and describe one of its distinguishing physical or behavioural characteristics (Table 6). There was a high correlation coefficient of 0.70 (p<0.001) between the number of years lived in the village and the age of a participant. Additionally gender and level of schooling were significantly correlated ($X^2 = 8.24$, df = 1, p = 0.004). Therefore, we only included the number of years a participant had lived in the village, and the participant's gender in our GLMM. This revealed that of participants living in areas where *A. lilacina* was likely to be present, men were more likely to be included in the better performing community data group than women (Coefficient value: 0.62 ± 0.31 (SE), p = 0.04), which is likely due to their spending more time outdoors in traditionally male working roles. The number of years a participant had lived in the community (Coefficient value: 0.012 ± 0.007, p = 0.14) had no significant effect.

Table 6: The gender, level of schooling, and mean number of years lived in the village, for all participants that lived in communities where parrots were likely to be present, and for those who were included in the best performing community data group (group three).

Response group	Gender	n	Level of schooling		Number of years in the village (mean)
			some	none	
All responses	Total	304	279	25	34
	Women	138	134	4	30
	Men	166	145	21	37
Community data group 3	Total	105	100	5	36
	Women	35	35	0	32
	Men	70	65	5	38

Discussion

We found that both field data and citizen science data in the form of community surveys were able to produce accurate species distribution models and their outputs had an overlap of 92%. When using field data, we found that models built using background points performed better than those built using absence points generated by eBird checklists, possibly due to the low frequency of eBird records in our study area. When using community data, we found the best performing models were those built using reports from observers who could name or recognise a photograph of *A.*

lilacina and correctly describe at least one distinguishing physical or behavioural characteristic.

Recent studies have shown that web-based citizen science projects and online biodiversity databases can be used to build reliable species distribution models (e.g. Saunders et al. 2020; Langham et al. 2015; Fournier et al. 2017). This study presents evidence that in areas where there are substantial barriers to web-based citizen science projects, for example in socio-economically deprived areas (e.g. Hobbs & White 2012), community surveys can overcome these barriers and produce accurate species distribution models. This is of particular use for newly described and rare species. Gender disassociation in local ecological knowledge is not uncommon (Kai et al. 2014; Aswani et al. 2018); we found that men were more likely to provide accurate answers than women and suggest that this is due to a gender difference in traditional working roles (Voeks 2007; Ayantunde et al. 2008) which allows men to spend more time outdoors. Erosion of local ecological knowledge is a global trend (Aswani et al. 2018) and we support the continuation of community wide engagement projects to minimise this risk, with a focus on support for women to enable them to engage with conservation.

After applying thresholds to our best performing field and community data models, they overlapped in their predictions of suitable habitat by 92% (in 9,314 km^2). The level of overlap we see between our community and field data models is greater than seen in similar comparison of eBird community data and field-based satellite tracking data of Band-tailed Pigeons *Patagioenas fasciata* (Coxen et al. 2017). Our community data model predicts a further 3,753 km^2 of suitable habitat that our field data model does not. These areas were closer to mangroves than areas predicted only by the field data model. This may be due to a factor of species detectability; *A. lilacina* are more detectable (highly vocal) when flying over to mangrove communal roost sites, so perhaps more likely to be seen by local communities in this habitat compared to when they are foraging inconspicuously in the dry forest (Ridgely & Greenfield 2001a). It is also possible that these areas represent locations in which local people have memories of the species occurring in the past, in which they no longer occur and thus were not recorded during field surveys. Our field data model predicts a further 4,655 km² of suitable habitat that our community data models do not, and in areas with a slightly higher mean annual NDVI than areas predicted only by the community model.

Similarly to Frey et al. (2013), we found variation in the accuracy of community data models built using different methods to filter interview responses. Our best performing model used a filter whereby participants needed to recognise a photograph of the species and provide a reliable description of how they distinguish it from other parrot species in their area. This suggests that, particularly in areas where many similar taxa may occur, the key to assessing the accuracy of information may be simply to ensure that participants are referring to the correct species. This draws parallels with checks that are in place for citizen science online databases such as eBird where records are flagged for systematic review and confirmed by a regional expert prior to their acceptance (Sullivan et al. 2014). It also supports the work of Frey et al. (2013) who conclude that, for easily-identifiable species at least, distribution modelling is possible using anecdotal reports. Our second best community data model (model one) greatly underestimated the predicted the area of suitable habitat. This group was based on the 'freelisting' method, where participants needed to name the parrot species in their area without any prior information or prompting. Previous studies using the freelisting method have yielded questionable results (e.g. Can & Togan 2009; Díaz-Ruiz et al.

2019) and we believe in our case, it was due to a very small sample size of participants who had the required natural history expertise to name this rare parrot species without any prompting or information.

We found that using identification of other parrot species, to measure overall biodiversity knowledge and therefore accuracy of answers, did not produce the most accurate results. This may be due to *A. lilacina's* unique daily migration behaviour, in some cases flying directly over villages and becoming conspicuous to many community members, not just those that are skilled at identifying multiple parrot species. Alternatively, it is possible that the two parrot species whose identification we assessed as a measure of reliability are incorrectly believed to be common and widespread throughout our study area (Ridgely & Greenfield 2001b; Freile & Restall 2018). Identification of other closely related species was not a good measure of data quality either in surveys investigating the distribution of a native pheasant species – results showed frequent misidentification of an 'imposter' pheasant photograph, but reliable information about the native pheasant was still generated (O'Brien et al. 1998).

Our distribution models based on field data and high quality community knowledge represent the first of their kind for the newly described and Critically Endangered *A. lilacina,* and have important conservation implications. With an estimated population size of just ~1,000 birds, and a suggested recent 60% population decline in parts of the range (Biddle et al. 2020), our results have identified new areas to survey. It is important to note that our model predictors did not include factors such as poaching that may have a strong impact on occupancy (Robinson et al. 2010). Whilst conducting community surveys for this study, we discovered a new large roost, unknown previously to local and international ornithologists, located near a socio-economically deprived coastal community, on a mangrove island. Even local residents, because of the conflict with pirates, deem this area as unsafe. We therefore recommend that when parts of a species range fall within areas that are rarely visited by outsiders, the combined knowledge of communities local to that species is likely to be much greater than that of external scientists or researchers, and should thus be used to enhance and supplement traditional field survey methods.

Conclusion and further research

Our distribution models highlighted an area of almost 18,000 km² where environmental conditions suggest the habitat is suitable for *Amazona lilacina*. Within these areas, there is variation in habitat type. The coastal region in general, compared to the rest of the country, has extremely little remaining remnant vegetation and only a small proportion of this is included in the national protected areas system (Cuesta et al. 2017). Furthermore, it is the most densely populated region of Ecuador and consequently represents a very anthropomorphically influenced landscape (CIESIN: Center for International Earth Science Information Network 2005). In the last ten years the Ecuadorian population has risen from 15 to 17.5 million, and there is no reason for this growth to slow (INEC 2010), but habitat modification by humans is a key factor threatening wild parrot populations worldwide (Snyder et al., 2000).

Amazon parrots have shown high levels of adaptability to human modified landscapes, with introduced populations being found worldwide as a result of the global pet trade (Mori et al. 2017; Uehling et al. 2019). Of the 36 Amazon parrot species, 14 have been reported outside of their natural range, and nine have established breeding populations (Mori et al. 2017). Although there are many factors involved in whether or not a species can become naturalised outside of its range, for example the popularity of the species in the pet trade and the frequency of accidental releases (Mori et al. 2017), the adaptability of a species is likely to play a part, with more specialised species being less able to survive. For example the Yellow-headed Amazon *A. oratrix*, is now found breeding in a park in Stuttgart, Germany, but arguably only because of the presence of the London Plane tree in which it exclusively nests (Martens & Woog 2017). In its native range this species has generalised nesting requirements, nesting in multiple tree species, but mainly dead palms - this lack of specialisation and ability to adapt to use cavities in the London Plane tree is likely to have contributed to its success in becoming naturalised outside of its traditional range (Eisermann 2003).

There are no reports of *A. lilacina* occurring outside of their natural range (Mori et al 2017). When considering a species with similar ecological needs to *A. lilacina;* the Red-tailed Amazon *A. brasiliensis* whose distribution is sympatric to the occurrence of mangroves and lowland forests (Martuscelli 1995), we also see no occurrence outside of its non-native range (Mori et al. 2017). It is likely that the complex habitat needs of these species restrict them from becoming naturalised in areas where these specific habitats do not occur. This specification may also reduce their ability to adapt to human modified landscapes and put them at a higher risk of extinction. In order to assess the level of habitat specificity of *A. lilacina* and identify any characteristics that may be associated with parrot presence, we conducted habitat assessments in areas that we know were suitable for parrots based on our observations and present the results of these in Chapter Four.

Appendices

Appendix 1

In all cases, prior verbal consent was obtained before conducting interviews and interviewees could decline from contributing once the purpose of the research was explained (in Spanish):

"We are collecting information about parrots as part of university research for Manchester Metropolitan University in the UK. The information gathered will be used to learn more about the parrots in this area and how we can help to conserve them, in their natural environment. I will be asking you some questions about the parrots you see around here, including where and how often you see them, but the answers will be completely anonymous. We will not be taking any names. We hope to publish this information in a scientific report, but the data will not be passed to others. You can decline to answer any questions you are not happy answering and you can withdraw at any time while completing the interview. Would you like me to repeat any information or do you have any further questions? Are you happy to take part in this research?
[If participant agrees to take part] Thank you for taking part. I will now begin the interview. [If participant declines consent to take part] Thank you for your time."

Appendix 2

All participants were asked the following questions by a local researcher in Spanish:

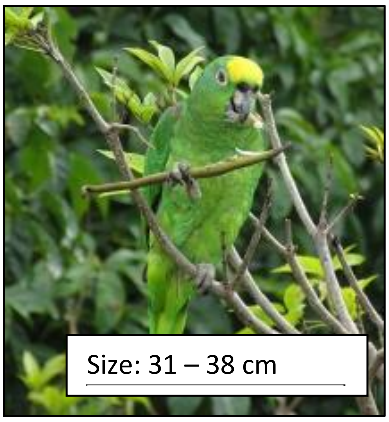
Section 1: demographic information (all participants)
Date: Name: Sex: Age: Level of schooling (none / primary / secondary / university): How many years have you lived in this village? GPS location (recorded by the interviewer):
Section 2: parrots in general (all participants)

Do you see parrots in this area? (yes / no / not sure): Can you name the parrots you see in this area? (yes / no / not sure): Of the following ten photographs, which do you see here? (yes / no / not sure): Do you see any other parrot species? (yes / no / not sure):
Section 3: *A. lilacina* specific (participants who report seeing *A. lilacina* in their area)
Where / in which habitat around here did you see *A. lilacina*? Approximately how many did you see? What were they doing? (flying / nesting / feeding / perched / something else) Do you see them all year round? (yes / no / not sure) How do you distinguish them from the other parrots? What kind of habitat do you think they prefer?

The following photographs were used to aid species recognition. Ten order combinations were available and rotated at random throughout the surveys. Most photographs used are taken from Athanas and Greenfield's 'Birds of Western Ecuador'. The photo of *Amazona lilacina* is by Daniel Arias and the photograph of the Brown-headed Parrot *Poicephalus cryptoxanthus* is by Bernard Dupont (cc: https://creativecommons.org/licenses/by-sa/2.0/deed.en).


Size: 38 – 41 cm


Size: 34 cm


Size: 31 – 38 cm

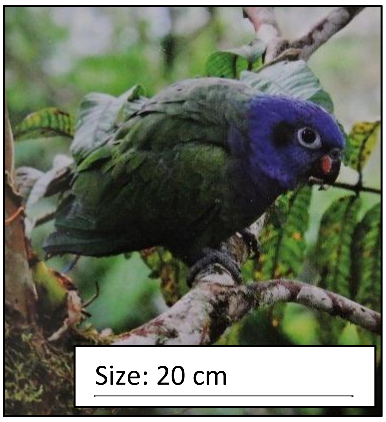

Size: 76 - 84 cm

Size: 20 cm

Size: 13 cm

Size: 28 cm

Size: 31 - 35 cm



Size: 20 cm

Size: 22 cm

Appendix 3

Evaluation metrics for all distribution models based on community data. Data were filtered based on different criteria (groups one to six). Models seven to seventeen are referred to as the combination models, with each representing a different combination of community data groups one to six. The mean and standard deviation of AUC and Tjur R^2 over five-fold cross validation are presented. The best performing model is built using community data group three (in bold).

Community Model	Sample size (after thinning)	Permutation importance of variables >10%	Mean of five-fold cross validation AUC (+/- sd)	Mean of five-fold cross validation Tjur R² (+/- sd)
1	27 (18)	Distance to mangrove (52%) Distance to river (14%) NDVI seasonality (20%)	0.83 (+/- 0.11)	0.48 (+/- 0.06)
2	155 (67)	Distance to mangrove (35%) Precipitation of wettest month (17%)	0.77 (+/- 0.10)	0.42 (+/- 0.03)
3	**115 (53)**	**Distance to mangrove (54%)** **Temperature seasonality (12%)**	**0.84 (+/- 0.05)**	**0.51** (+/- **0.01)**
4	134 (64)	Distance to mangrove (43%) Precipitation of wettest month (22%)	0.76 (+/- 0.06)	0.41 (+/- 0.01)
5	67 (40)	Distance to mangrove (53%)	0.76 (+/- 0.07)	0.40 (+/- 0.03)
6	40 (26)	Distance to mangrove (37%) Precipitation of wettest month (19%)	0.74 (+/- 0.03)	0.39 (+/- 0.02)
7 (3+4+5+6)	28(18)	Mean diurnal temperature range (54%) Distance to mangrove (30%)	0.80 (+/- 0.04)	0.44 (+/- 0.06)
8 (3+4+5)	40 (26)	Distance to mangrove (34%) Annual mean NDVI (26%) Mean diurnal temperature range (20%)	0.80 (+/- 0.07)	0.51 (+/- 0.02)
9 (3+4+6)	67 (37)	Distance to mangrove (48%) Precipitation of the coldest quarter (13%)	0.82 (+/- 0.07)	0.48 (+/- 0.04)
10 (3+5+6)	30 (20)	Distance to mangrove (60%) Mean diurnal temperature range (15%)	0.74 (+/-0.13)	0.49 (+/- 0.03)
11 (4+5+6)	39 (24)	Distance to mangrove (44%) Mean diurnal temperature range (33%)	0.78 (+/- 0.07)	0.43 (+/- 0.04)
12 (3+4)	102 (50)	Distance to mangrove (56%) Temperature seasonality (12%)	0.82 (+/- 0.04)	0.50 (+/- 0.01)
13 (3+5)	45 (29)	Distance to mangrove (51%) Mean diurnal temperature range (11%) Annual mean NDVI (10%)	0.78 (+/-0.08)	0.52 (+/- 0.03)
14 (3+6)	76 (40)	Distance to mangrove (57%) Annual mean temperature (15%)	0.81 (+/-0.04)	0.49 (+/- 0.03)
15 (4+5)	54 (34)	Distance to mangrove (467%) Mean diurnal temperature range (23%) Annual mean NDVI (11%)	0.75 (+/- 0.06)	0.41 (+/- 0.03)
16 (4+6)	90 (48)	Distance to mangrove (39%) Precipitation of the coldest quarter (21%) Precipitation of the wettest month (12%)	0.73 (+/- 0.08)	0.41 (+/- 0.01)
Community model 17 (5+6)	45 (27)	Distance to mangrove (46%) Precipitation of the coldest quarter (16%) Precipitation of the driest month (15%)	0.69 (+/- 0.05)	0.39 (+/- 0.04)

Chapter Four: The fine-scale habitat preferences of *Amazona lilacina*

Abstract

Parrots show a wide range in adaptability and ecological specialisation, with some surviving in towns and cities, whilst other remain restricted to areas where certain ecosystems occur. Human modification of landscape is a global threat to wildlife and the ability to adapt to such a changing landscape is likely to affect species extinction risk. The newly described *Amazona lilacina* occurs in the lowland coastal regions of Ecuador, the most densely populated region of the country, but we know little of its habitat preferences. We studied four areas where it occurs, including a protected area (CB2) and the disturbed boundary of this protected area (CB1), a community owned forest (LB) and recovering agricultural land (IS). We successfully observed parrots in all areas apart from the recovering agricultural land. We collected data on landscape and local scale habitat characteristics in the 200 m^2 surrounding presence ($n = 42$) and 'pseudo-absence' ($n = 132$) points. We compared the vegetation characteristics between four areas using Kruskal Wallis tests, and between presence and absence points using Wilcoxon rank sum tests. We then used random forests to assess the ability of habitat characterises to distinguish between parrot presence and absence. We recorded the species using five trees for nesting, four for roosting and at least 33 for feeding, but found that the habitat variables across the four areas were significantly different. We suggest the area within a privately protected reserve is most suitable, due to its significantly lower evidence of humans, higher evidence of wild animals, and greater quantity and species richness of *A. lilacina* feeding trees. In the morning, parrots preferred areas at high elevation, with larger but fewer trees, whilst in the day

when foraging, they preferred areas at significantly lower elevation with larger and more trees. This suggests that they require a variety of habitat types in order to fulfil their daily requirements. Our random forest analysis was able to correctly distinguish absence points for the three areas but had a lower level of accuracy in predicting presence points. The predict function of random forests suggested that recovering agricultural land was suitable for parrots based on data from the disturbed protected area and community owned land, but not suitable based on data from the undisturbed protected area. This reflects the large difference between areas that parrots use, and thus highlights the need for landscape scale conservation to preserve the mosaic of habitats they need.

Introduction

Habitat availability is a limiting factor for the survival of many species worldwide and habitat loss has been linked directly to species extinction (Brook et al. 2003). A third of all Neotropical parrots are threatened with extinction and half of all populations are affected by habitat loss caused by agriculture and logging (Berkunsky et al. 2017; BirdLife International 2018c). *Amazona lilacina* is a Critically Endangered parrot endemic to the coastal provinces of Ecuador where it occurs over a patchwork landscape of dry forest, mangrove and agricultural land (Biddle et al. 2020; BirdLife International 2020b). One of its key threats is habitat loss and alteration as a result of expansion of the human population (Ridgely & Greenfield 2001a). Ecuador has been ranked as the sixth highest global priority for parrot conservation (Olah et al. 2016). Rapid deforestation for agriculture and anthropomorphic expansion led to the loss of half its original forest cover by 2008 (Mosandl et al. 2008). The lowland areas have been most intensively cleared (Tapia-Armijos et al. 2015) and as a result it is estimated that just 1% of dry forests (Dodson & Gentry 1991) and 60% of mangroves remain

(Hamilton 2013). Consequently, these ecosystems are listed as global conservation priorities due to their high biodiversity value and high level of threat from human activity (Miles et al. 2006; Romañach et al. 2018).

A detailed knowledge of species ecological requirements and how these are met by their habitat is essential in order to understand the possible effects of habitat loss or alteration, and species with specialised ecological requirements may be more at risk of extinction than others (Frank & Amarasekare 1998; Davies et al. 2004). For example, it is suggested that cavity nesting birds, such as parrots, are affected by logging more than other species, because previously or selectively logged forests have significantly fewer and less suitable nesting cavities, than non-logged areas (Politi et al. 2010; Costantini et al. 2016; De Labra-Hernández & Renton 2016). Parrots also rely on species such as woodpeckers to initiate cavity creation (Guix et al. 1999), and these occur at a lower density and diversity in previously logged areas (Styring & Ickles 2001). Habitat alteration and logging may also affect foraging behaviour. A review of parrot densities and abundance worldwide, also suggests that parrot density is lower outside of primary or secondary forest (in selectively logged, fragmented, monocultural plantations) which suggests that they are having to use larger areas (Marsden & Royle 2015). Indeed, it has been shown that Amazon parrots forage over an area ten times as large when resource is limited, which is likely to have an energetic cost to the individual (Salinas-Melgoza et al. 2013).

Amazona lilacina occur only in the lowlands of Ecuador, up to 700 m above sea level (Ridgely & Greenfield 2001a; Forshaw & Knight 2010; Freile & Restall 2018). They are reported to inhabit deciduous forest and mature secondary woodlands (Ridgely & Greenfield 2001a), scrubby dry forests and light woodlands (Juniper & Parr 1998; Athanas & Greenfield 2016), forest canopy, and forest edges (Freile & Restall 2018).

They roost in mangroves (Ridgely & Greenfield 2001a; Athanas & Greenfield 2016; Freile & Restall 2018) or more rarely stands of gallery trees (Forshaw 1981), and they nest in cavities formed in the trunks of large soft wood trees (Ridgely & Greenfield, 2001). Site protection and management are the two most urgently and frequently required conservation measures for Neotropical parrots (Olah et al. 2016; BirdLife International 2018c). Following a recent conservation status review, *A. lilacina* has been up-listed on the IUCN Red List to Critically Endangered (Biddle et al. 2020; BirdLife International 2020a) however, the details regarding its specific ecological needs are unclear. Therefore, in order to help understand the species ecology and assist with conservation planning, this study aims to:

i. confirm which tree species are required for feeding, nesting and roosting;
ii. compare the habitat characteristics between areas used and not used by *A. lilacina* to distinguish key characteristics associated with presence;
iii. predict the suitability of habitat in an area in which *A. lilacina* are believed to occur but species observations were not collected during this study.

Methods

Study area

Four areas were included in this study (Figure 1) representing different habitat types, a protected area, a disturbed protected area, community owned land, and recovering agricultural land. The protected area, the Cerro Blanco is ~6,000 hectares of dry tropical forest managed by Fundación Pro Bosque, a non-profit organisation whose main objective is to protect the forest from threats such as fire, hunting, land trafficking and the development of squatter settlements, through ranger presence, reforestation, research, environmental education and ecotourism (Horstman 2017). As a dry forest, it is characterised by 50% drought-tolerant deciduous trees, a mean annual temperature of >25ºC, and rainfall of 700 - 900 mm per year falling mainly between January and April (Sánchez-azofeifa et al. 2005; Horstman 2017). This is an area of forested hills, with little human presence, used by *A. lilacina* for nesting and foraging (R. Biddle pers. obs.). The boundary of this protected area is much more disturbed, with a network of buildings, trails and trees are accessible to paying visitors. Despite disturbance, this area is also used by *A. lilacina*, but mainly in the morning as the birds arrive from their El Salado mangrove roost site.

The remaining two areas in this study represent places where little is known about the habitat use of *A. lilacina*. Las Balsas is community owned land ~50 km north-west of the Cerro Blanco Forest, in the Cordillera Chongón Colonche. This population of *A. lilacina* has been known by the local community for at least 50 years but was not recorded in any literature or online species databases until 2016 (Blanco et al. 2016). It occurs far from the coast and is the only identified population that does not use mangroves to roost, instead existing across a patchwork of community owned forest and agricultural land. Isla Santay is an island in the big city of Guayaquil, made up of coastal mangrove, shrubby dry forest, agriculture and recovering agricultural land. It is listed as a recreational area under Ecuador’s National System of Protected Areas (NSPA), which means that since 1990 no further settlement or agriculture has been permitted. The island is managed by the Ministry of Environment (MAE) and accessible to paying visitors by a pedestrian bridge from Guayaquil. There have been reports of *A. lilacina* foraging on the island from MAE wardens, and we have observed them in the coastal mangroves.

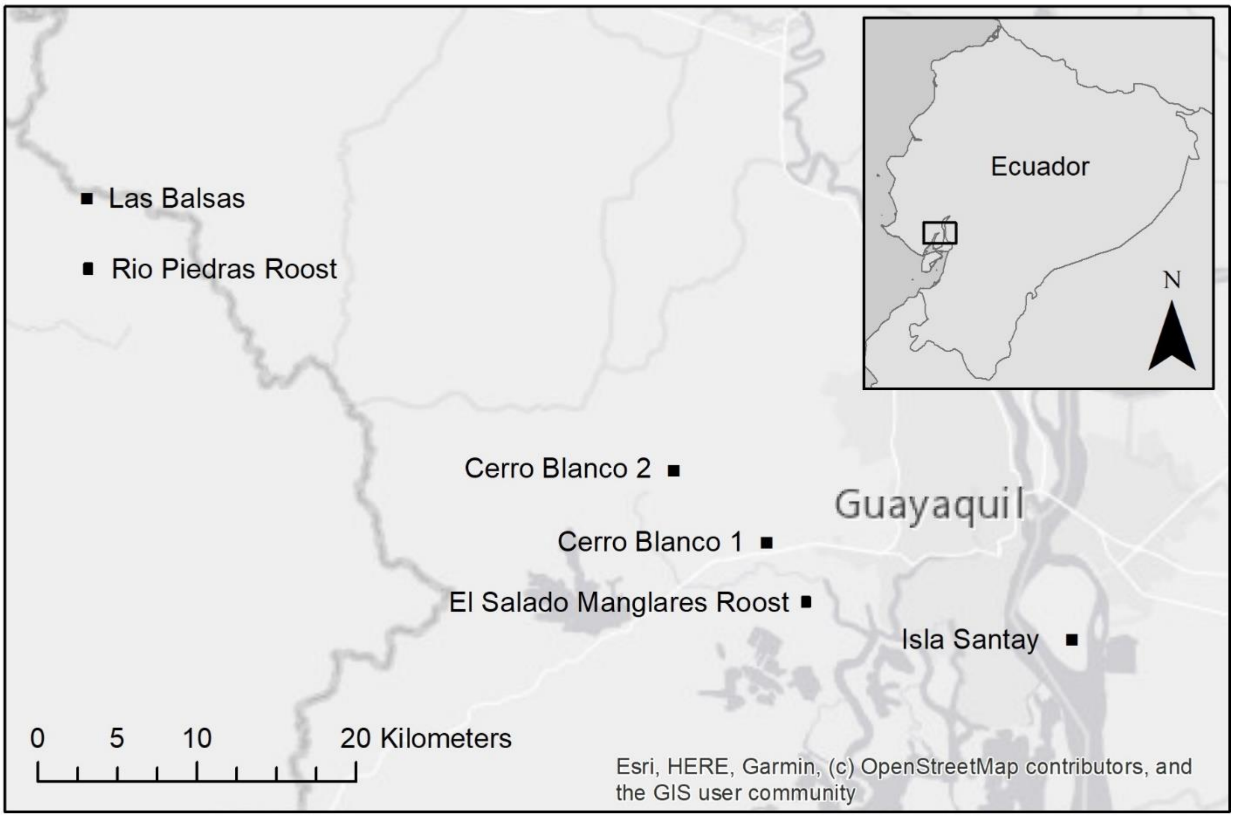


Figure 1: The four dry forest study areas were; Cerro Blanco 1 (CB1) the disturbed protected area, Cerro Blanco 2 (CB2) the protected area, Las Balsas (LB) the community owned land and Isla Santay (IS) the recovering agricultural land. The two nearby communal roosting sites for *Amazona lilacina* are also shown; El Salado and Rio Piedras.

Parrot presence and absence points

Between September 2016 and February 2017, we collected georeferenced locations of parrot presence and parrot absence points using Garmin eTrex GPS units. Our presence points referred to the exact location where we had seen parrots perched, and our 'pseudo-absence' points referred to locations within the vicinity of parrots (<2 km from parrot sightings) but where no observations of parrots were made during our study. Due to the secretive nature of this species whilst foraging, we devised two methods to observe parrots. First, we occupied vantage points over roost sites to visually track birds arriving into the forest between 05h00-08h00, projected the georeferenced locations of the parrots, and then later ground-truthed those locations. Second, we conducted forest transects along ranger patrolling routes between 08h00-11h00 and 14h00-18h00 in small groups, whilst continuously scanning the canopy and quietly listening for parrots. Presence points were collected successfully in three of the four study areas, and our aim was to compare these with absence points in order to identify any habitat characteristics that parrots may prefer. To collect absence points, we followed a systematic approach whilst walking forest transects, scrambling 15 m into the forest on alternate sides, every 100 m. The transects followed paths that are used as foot access into the forest and between the guard stations in the protected area. These stations are based a high locations, and therefore the trails roughly follow the high ridges of the Cerro Blanco (white hill). It was not possible to select random locations due to the inaccessibility of the forest during the rainy season, and the desire not to cut paths or damage reforested and regenerating areas.

Habitat characteristics

To record habitat characteristics associated with parrot presence and absence, we formed small groups consisting of two Chester Zoo staff and one ranger from Fundación Pro Bosque. We treated each presence and absence point as a central point from which we walked 10 m in each compass direction to mark out a large quadrat of 200 m^2 (Figure 2). We waited until times of low bird activity to collect data, in order to minimise disturbance to parrots, and completed each quadrat in approximately 30 minutes. We followed similar methods to previous studies assessing the effect of habitat characteristics on bird presence / abundance (Chettri et al. 2005;

Reid et al. 2012; Zhang et al. 2013) and recorded landscape-scale variables of elevation (m) and XY coordinate, and the following local patch-level characteristics:

- number of trees with a diameter at breast height (dbh) of >30 cm;
- combined total dbh (cm) for the quadrat area;
- for trees with a dbh >30cm;
    - exact dbh (cm);
    - species and if this is used by *A. lilacina* for feeding, roosting or nesting, based on our observations, observations of the rangers including across different seasons, and observations reported by other researchers (Kunz, 1996; Juniper & Parr, 1998; Berg & Angel, 2006; Blanco et al., 2016);
- density of foliage at chest height, measured by the number of squares in a clear plastic grid (10 cm by 10 cm) that are not obscured by leaves or woody material when looking from the central point to each corner of the quadrat, then averaged;
- level of canopy openness, measured by the number of squares in the same clear plastic grid, that are not obscured leaves when looking upwards at the central point;
- percentage ground cover in the categories of exposed soil, leaf litter, herbaceous plants (plants ≤1 m) or shrubs (plants ≥1 m);
- signs of animal biodiversity as an indicator of ecosystem health, for example seen or heard specimens (mainly invertebrates, birds, reptiles), droppings or feeding marks - recorded as yes or no;
- signs of indirect or direct human disturbance such as signs of fire, cut wood and logging, rubbish, grazing animals or dogs - recorded as yes or no.

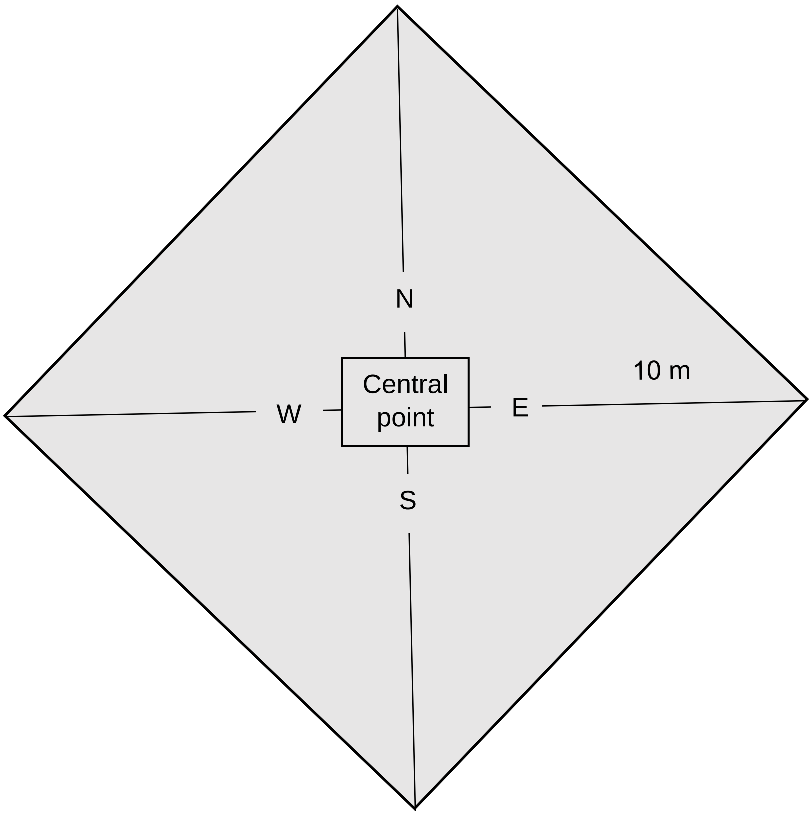


Figure 2: A 200 m² quadrat was marked out around the central point which represents a parrot presence or absence point, by walking 10 m in each compass direction. Within this area, habitat characteristics were recorded.

Data analysis

To make an initial comparison between the forest types in the four areas we used habitat variables collected at the absence points. We calculated median and interquartile range for each and used Kruskal Wallis tests to identify significant differences. In order to compare between the parrot presence and absence points, within the three areas for which these data were collected, we conducted unpaired two-sample Wilcoxon tests in R using the function 'wilcox.test'.

In order to assess the ability of the habitat variables to distinguish between parrot presence and absence in each area independently, and then for all areas combined, we conducted random forests analysis (Breiman 2001) using the R package ‘randomForest’ (Breiman et al. 2018). This analysis uses an ensemble of decision trees to classify data, with each tree predicting a classification, and the classification with the most votes becoming the model prediction. For each random forests model, we present the confusion matrix which confirms how many of the presence and absence points were correctly or incorrectly classified, and two model comparison methods; the out of bag error rate (OOB) and the mean decrease in accuracy (MDA). Random forests uses bagging (bootstrap aggregation), whereby a subset of the data is used to build the model, and then the remaining data (out of bag) is used as a test sample. The out of bag error rate (OOB) therefore measures the ability of the model to correctly classify the out of bag data, and provides a robust method of evaluating each model’s predictive performance (Breiman 2001). The mean decrease in accuracy (MDA) is presented for each variable, within each random forest model. This value represents the difference in predictive accuracy for the out of bag data when a variable is removed, and so the higher the value the more important that variable is for successful classification between presence and absence (Cutler et al. 2007).

For the recovering agricultural areas, only absence data exists. To determine whether this area is suitable for parrots or not we used the predict function of the random forest package, to classify each location in the Isla Santay recovering agricultural area, based on the models built using presence and absence data from the protected area, the disturbed protected area, the community owned land, and then combined. All data analysis was conducted in R version 4.0.2 (R Core Team, 2020).

Results

Key feeding, roosting and nesting tree species

We collected data from a total of 179 locations; 42 parrot presence points and 137 parrot absence points (Table 1). During the study, parrots were observed using the protected area, the disturbed protected area, and the community owned land. In the past, parrots have been seen using Isla Santay the recovering agricultural land - however, during our study no parrots were seen here. Across all sites, we recorded 60 tree species during habitat assessments, and with the help of Fundación Pro Bosque plant nursery staff, were able to identify most of these to species level. Based on our observations, those of the forest rangers, and previously published literature (Kunz, 1996; Juniper & Parr, 1998; Berg & Angel, 2006; Blanco et al., 2016) we confirm that *A. lilacina* feeds on the flowers, fruits or seeds of 33 species, and roosts in either mangrove or Algarrobo trees. Active nests have been documented in Ceibo, Pigio, Bototillo, Guarumo and Balsa (Table 2).

Table 1: Habitat variables were collected at 179 locations across four areas that parrots use.

	Area				Total
	Disturbed protected area (CB1)	Protected area (CB2)	Community owned land (LB)	Recovering agricultural land (IS)	
Total points	17	122	24	16	179
Presence points	10	23	9	0	42
Absence points	7	99	15	16	137

Table 2: Tree species used by *A. lilacina* for feeding, nesting and roosting.

Local Spanish name	Scientific binomial	Family	Feed	Nest	Roost
Pigío	*Cavanillesia platanifolia*	*Malvaceae*	Feed	Nest	
Ceibo	*Ceiba trichistandra*	*Malvaceae*	Feed	Nest	
Balsa	*Ochroma pyramidale*	*Malvaceae*	Feed	Nest	
Bototillo	*Cochlospermum vitifolium*	*Bixaceae*	Feed	Nest	
Guarumo	*Cecropia litoralis*	*Urticaceae*	Feed	Nest	
Mangle jelí	*Conocarpus erectus*	*Combretaceae*			Roost
Mangle Blanco	*Laguncularia racemosa*	*Combretaceae*			Roost
Mangle negro	*Avicennia germinans*	*Acanthaceae*			Roost
Algarrobo	*Prosopis julifora*	*Fabaceae*	Feed		Roost
Cocobolo	*Cynometra bauhiniifolia*	*Fabaceae*	Feed		
Guarango	*Caesalpinia spinosa*	*Fabaceae*	Feed		
Beldaco	*Pseudobombax millei*	*Malvaceae*	Feed		
Cascol	*Caesalpinia glabrata*	*Fabaceae*	Feed		
Ciruela	*Spondias purpurea*	*Anacardiaceae*	Feed		
Ficus	*Ficus sp*	*Moraceae*	Feed		
Guaba	*Inga spectabillis*	*Fabaceae*	Feed		
Guanabana	*Annona muricata*	*Annonaceae*	Feed		
Guasmo	*Guazuma ulmifolia*	*Malvaceae*	Feed		
Guayaba	*Psidum sp*	*Malvaceae*	Feed		
Higueron	*Ficus membranaceae*	*Moraceae*	Feed		
Mango	*Mangifera indica*	*Anacardiaceae*	Feed		
Matapalo	*Ficus cuatrecasana*	*moraceae*	Feed		
Muyuyo	*Cordia lutea*	*boraginaceae*	Feed		
Naranjo	*Citrus sinensis*	*Rutaceae*	Feed		
Pechiche	*Vitex gigantea kunth*	*Lamiaceae*	Feed		
Pela Caballo	*Leucaena trichodes*	*Fabaceae*	Feed		
Pepito Colorado	*Erytrina velutina*	*Fabaceae*	Feed		
Guachapeli	*Pseudosamanea guachapele*	*Fabaceae*	Feed		
Ciruelo de Monte	*Spondias mombim*	*Anacardiaceae*	Feed		
Cojojo	*Acnistus arborescens*	*Solanaceae*	Feed		
Erythrina	*Erythrina fusca*	*Fabaceae*	Feed		
Tutumbo	*Cordia eriostigma*	*Boraginaceae*	Feed		
Tinto	*Lonchocarpus apurpurea*	*Fabaceae*	Feed		
Vanillo	*Senna sp.*	*Fabaceae*	Feed		
Palo Santo	*Bursera graveolens*	*Burseraceae*	Feed		
Bolsa de chivo	*Stemmadenia obovata*	*Apocynaceae*	Feed		

Comparison of areas used by parrots

When comparing the absence points across the four areas, we see significant differences in 12 of the 15 habitat variables (Table 3). For instance, the disturbed protected area and the recovering agricultural land, are at much lower elevation than the other two areas. In the protected area, the foliage is considerably denser at chest height than the other three areas and it has a much higher mean percentage of leaf litter cover. In comparison, the recovering agricultural land has as a considerably more open canopy than the other three areas, fewer larger trees, and therefore a much lower total tree diameter at breast height. There is also a difference in the number of tree species that are important for parrots; the protected area has up to six feeding and one nesting tree per quadrant, but no roosting tree species which are only present in the recovering agricultural area and the community owned land. Finally, there are differences in the level of human disturbance, with 100% of quadrats in the disturbed protected area showing some evidence of human disturbance, 88% in the recovering agricultural land, 47% in the community owned land, and 11% in the protected area.

Table 3: The lower quantile, median and upper quantile of each habitat variable at the absence and presence points in each of the four areas. The results of Kruskal Wallis tests indicate the ‘difference between the four areas’ and p-value results from Wilcoxon tests indicate ‘difference between presence and absence’ within each area. CB1 is the disturbed protected area, CB2 the protected area, LB the community owned land and IS the recovering agricultural land.

Variable	Value	Absence				Difference between four areas		Presence			Difference between presence and absence (p value)		
		CB1	CB2	IS	LB			CB1	CB2	LB			
		N = 7	N = 99	N = 15	N = 16	H	P	N = 10	N = 23	N = 9	CB1	CB2	LB
Elevation (m)	LQ	24	350	10	429	67	<0.001	63	112	84	<0.001	<0.001	0.049
	MED	24	365	11	450			68	327	120			
	UP	37	379	13	459			71	343	370			
Chest height density (%)	LQ	54	29	47	65	26	<0.001	48	31	45	0.696	0.377	0.355
	MED	70	43	60	78			60	47	59			
	UP	84	60	84	91			87	74	80			
Canopy (%)	LQ	31	8	35	10	13	0	76	9	35	0.256	0.665	0.134
	MED	50	30	100	20			85	18	45			
	UP	65	67	100	45			95	50	90			
Leaf litter (%)	LQ	10	30	0	5	29	<0.001	53	35	0	0.022	0.657	0.038
	MED	10	60	0	30			80	50	0			
	UP	30	90	8	95			90	70	20			
Herb layer (%)	LQ	30	0	9	0	38	<0.001	0	0	0	0.002	0.023	0.307
	MED	45	0	35	0			0	0	20			
	UP	48	7	91	30			0	0	30			
Bare soil (%)	LQ	0	0	0	0	3	0.41	3	0	5	0.110	0.44	0.224
	MED	0	0	0	0			10	0	10			
	UP	8	5	0	10			14	3	20			

Shrub layer (%)	LQ	25	2	4	5	3	0.43	0	18	20	0.009	0.100	0.322
	MED	35	13	25	30			0	45	50			
	UP	45	50	80	49			9	50	60			
Number large trees	LQ	5	3	0	4	18	<0.001	2	5	2	0.223	0.044	0.187
	MED	8	5	2	7			3	8	4			
	UP	9	9	4	9			4	10	6			
Largest tree (cm)	LQ	50	55	23	60	23	<0.001	64	71	43	0.013	0.038	0.107
	MED	53	81	31	69			107	113	50			
	UP	70	117	44	86			124	176	63			
Total tree dbh (cm)	LQ	224	172	23	274	21	<0.001	123	290	81	0.045	0.002	0.084
	MED	287	320	68	349			149	590	152			
	UP	330	490	168	446			177	743	345			
Number of feeding trees	LQ	1	1	0	0	26	<0.001	0	2	0	1	0.649	0.424
	MED	1	3	0	1			1	3	0			
	UP	1	6	1	5			2	6	2			
Number of nesting trees	LQ	0	0	0	0	14	0	0	0	0	0.142	0.474	0.491
	MED	0	0	0	0			0	1	0			
	UP	0	1	0	0			1	1	0			
Number of roosting trees	LQ	0	0	0	0	43	<0.001	0	0	0	NA	NA	1
	MED	0	0	0	0			0	0	0			
	UP	0	0	1	1			0	0	1			
Human evidence (1=yes, 0=no)	LQ	1	0	1	0	61	<0.001	0	0	1	0.016	0.329	0.228
	MED	1	0	1	0			0	0	1			
	UP	1	0	1	1			1	1	1			
Animal evidence (1=yes, 0=no)	LQ	0	0	0	0	7	0.079	0	0	0	0.954	0.005	0.152
	MED	0	0	0	0			0	0	0			
	UP	1	0	0	0			1	1	0			

Comparison of presence and absence points

When comparing areas of parrot presence with absence, in the disturbed, protected area and community owned land, there were significant differences in a number of forest characteristics (Table 3) but the only one that was significant across all areas was elevation (Figure 3). Parrots use locations at lower elevation in the protected areas and the community owned land, but locations at higher elevation in the disturbed protected area (Figure 3). Throughout Cerro Blanco parrots use locations where the size of the largest tree is significantly bigger than in areas they do not use. In the protected areas, this is coupled with significantly higher leaf litter cover and lower herb and shrub layer cover, whilst in the disturbed protected area on the boundary of the forest, there are no significant differences in ground cover. In the protected area, parrots use locations with a greater number of large trees, and therefore a bigger overall diameter at breast height, whereas in the disturbed protected area, parrots select for areas with significantly fewer large trees (three rather than seven).

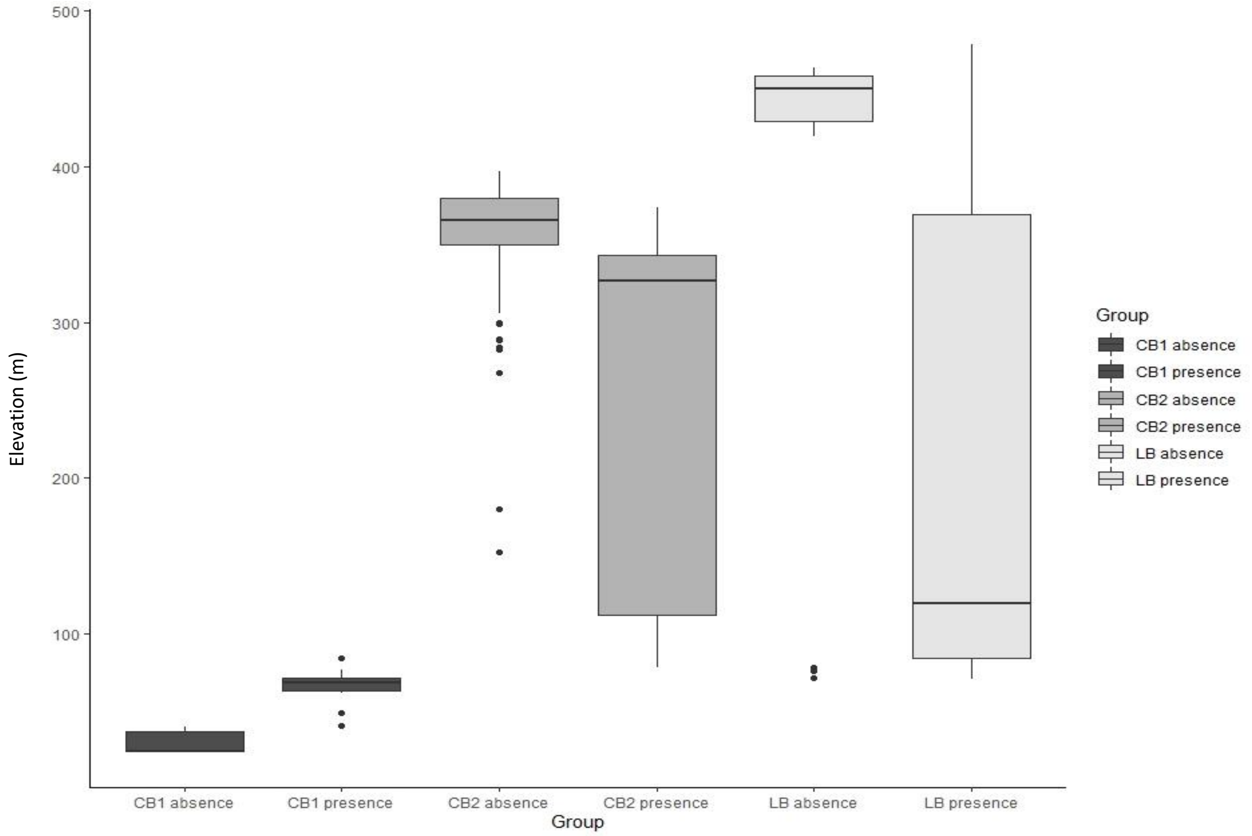


Figure 3: Boxplots to show the range of elevation (m) in areas of parrot presence and absence. CB1 is the disturbed protected area, CB2 the protected area, LB the community owned land and IS the recovering agricultural land.

Classification of presence and absence areas

Random forest analysis was able to distinguish between presence and absence points in the disturbed protected area with a high accuracy (OOB error rate: 6%); all of the absence points were categorised correctly, as were nine of the ten presence points. When looking at data from the protected area, models had lower accuracy (OOB error rate: 11%); but again were able to predict all of the absence points correctly, but only ten of the 23 presence points. Of those that were incorrectly predicted, i.e. those that showed a value of <0.5 for predicted probability presence (n = 14), the mean predicted presence value was 0.26 (+/- 0.15). For the community owned land, models had an even lower accuracy (OOB error rate: 17%), correctly predicting all of the absence points (15) but just five of the nine presence points. When combining the data across the three areas, models showed an error rate of 16%, and were able to correctly classify 135 of the 137 absence points but again just 16 of the 42 presence points (Table 4).

Table 4: Confusion matrix showing the out of bag error estimates and classification results of random forest models for the disturbed protected area (CB1) the protected area (CB2) and the community owned land (LB). Numbers in bold indicate where models are able to correctly distinguish between presence and absence.

CB1 - disturbed protected area			
OOB estimate of error rate: 6%	predicted presence	predicted absence	Error
Actual presence (n = 10)	**9**	1	0.1
Actual absence (n = 7)	0	**7**	0.0
CB2 - protected area			
OOB estimate of error rate: 11%	predicted presence	predicted absence	Error
Actual presence (n = 23)	**10**	13	0.56
Actual absence (n = 99)	0	**98**	0.01
LB - community owned land			
OOB estimate of error rate: 17%	predicted presence	predicted absence	Error
Actual presence (n = 9)	**5**	4	0.44
Actual absence (n = 15)	0	**15**	0
All areas combined			
OOB estimate of error rate: 16%	predicted presence	predicted absence	Error
Actual presence (n = 42)	1**6**	26	0.62
Actual absence (n = 137)	2	**135**	0.01

Across all areas, elevation had the highest mean decrease in accuracy value (MDA) and was therefore most powerful at distinguishing between parrot presence and absence locations (Table 5). The second and third most important variables associated with the presence of parrots differs for each area: for the disturbed protected area it is herb layer and shrub layer cover, for the protected area it is number of large trees and shrub layer, and for the community owned land it is the number of nesting and roosting trees (Table 5).

Table 5: The mean decrease in accuracy for random forests models for each area. Values give an indication of how important that variable is in distinguishing between presence and absence points.

Variable	Mean decrease in accuracy			
	Disturbed protected area (CB1)	Protected area (CB2)	Community owned land (LB)	All areas combined
Elevation	10.21	20.98	13.48	26.23
Density at chest height	-1.62	-0.38	0.02	1.88
Canopy	2.95	1.97	-1.46	8.33
Leaf litter	3.82	4.96	5.92	7.82
Herb layer	9.90	1.15	1.23	11.09
Soil exposed	-0.88	3.47	4.65	7.66
Shrub layer	5.83	6.76	2.32	6.14
Number of large trees	2.70	8.09	2.54	8.45
Total tree dbh	2.12	6.67	2.50	12.66
Largest tree dbh	3.02	4.14	3.89	9.15
Number of feeding trees	-2.12	1.59	5.19	5.55
Number of nesting trees	1.34	1.88	0.00	1.24
Number of roosting trees	0.00	0.00	5.66	5.02
Human evidence	2.73	3.04	-1.19	7.20
Animal evidence	-0.37	0.53	0.00	-1.20

We used the random forest models built with data from the three areas in which parrots were seen, to predict which of the 16 locations within the recovering agricultural land in Isla Santay may be suitable for parrots. All 16 locations were predicted to be presence points. On the contrary, when using the model built with data from just the protected area, 14 of the 16 recovering agricultural locations were predicted as absence points. When using the model built with just community owned land data, all 16 recovering agricultural locations are predicted to be presence points, and again on the contrary, when using the random forest model built using all the data combined, all 16 of the recovering agricultural land locations were predicted to be absence points (Table 6).

Table 6: Predictions from random forest models built with data from each area, to show whether the locations in recovering agricultural land (IS) are suitable for parrots.

Random forest model built using data from:	Random forest predictions for the 16 IS locations	
	Predicted presence	Predicted absence
CB1 disturbed protected area	16	0
CB2 protected area	2	14
LB community owned land	16	0
Combined CB1, CB2 and LB data	0	16

Discussion

This study has shown that *A. lilacina* are present in at least three areas with significantly different fine-scale habitat characteristics, where they are known to use just five tree species for nesting, four for roosting and 33 for feeding. Our results suggest that the forest within a privately protected reserve, is likely the most suitable area for the species, because it has the most feeding and nesting trees, and the least evidence of humans which are known to pose threats to the species' survival (Horstman 2017). Across all areas we found significant differences between the locations that are used, and not used, by *A. lilacina* suggesting that similarly to other Amazon parrots, they have a clear preference for certain habitats and characteristics (Gilardi & Munn 1998; Salinas-Melgoza et al. 2013; Figueira et al. 2015). These preferences vary depending on the area, for example between the disturbed protected area, which we observed being used predominantly at sunrise, and deeper inside the protected area which was used predominantly during the daytime. We suggest that these areas have different functions and that *A. lilacina* requires a mosaic of habitats, and has relatively complex habitat needs which may put it at an increased risk of extinction (Owens & Bennett 2000). Our models provided vastly different predictions regarding whether or not a recovering agricultural area was suitable for parrots, which further emphasises the variation in habitats this species requires.

We found that elevation is the most important contributing factor for distinguishing between presence and absence in all areas, but in the morning there is preference for higher elevation, whilst in the daytime the preference is for lower elevation. Overall, our results suggest that when birds are arriving from their roost site they prefer a higher elevation with one large dominating, whilst in the day when foraging they prefer a lower elevation and area with multiple large trees. In the forests of Peru, Amazon parrots were also found to avoid areas of high ground when foraging whilst larger bodied parrots, i.e. macaws, showed a preference for high ground (Gilardi & Munn 1998). This perhaps reflects *A. lilacina's* secretive and cautious nature when foraging and vulnerability to avian predators such as the Grey-backed Hawk *Pseudastur occidentalis* (Kunz 1996; Ridgely & Greenfield 2001a).

We found that tree size was significantly different in locations used by parrots compared to those that were not, in both the protected and the disturbed protected areas. In the disturbed protected area, the diameter at breast height (dbh) of the largest tree was seven times larger in areas used by parrots, and in the protected area it was over twice as large. This association is not found for other large frugivores (macaws, toucans, curassows, guans, chacalacas) in the forests of western Ecuador (Walter et al. 2017) which suggests that for Amazon parrots, these large trees may play a particularly important role, not just for feeding. In the disturbed protected area we found parrots using areas with fewer large trees, one of which was extremely large, but in the protected area, they had a preference for areas with a greater density of large trees. The protected area was used mainly by birds in the day for foraging, so this preference may be associated with selection of larger trees more mature trees that have a greater feeding resource. We also found that in this protected area, parrots use locations with significantly more evidence of other native animal species, which again is perhaps associated with a more mature forest (Lawton et al. 1998). It is suggested that nesting trees for Amazon parrots must be a minimum of 59 cm dbh (Snyder et al. 1987), however we believe it is unlikely that parrots are selecting for areas with nest trees as they are known to be extremely cautious around nest cavities if observers are present (Enkerlin-Hoeflich et al. 2006).

Our models to classify between presence and absence in each area independently had relatively low error rates (< 17%). However, the predictions of these models into a new area of recovering agricultural land in which we gathered no species observations, were vastly different. They suggested this area is suitable for *A. lilacina* based on information from community owned and disturbed protected forest, but not suitable for parrots based on information from a protected area. This highlights the large variation between the three areas that we observed parrots in, and indicates that there may be additional predictors of parrot presence that we did not capture in our methods. Our data collection protocol was time efficient and easily replicated by multiple members of staff speaking different languages, and our focus was on identifying the trees that are important to this species. Now this list has been compiled, future studies should pay particular attention to the phenology of these species whilst conducting habitat assessments. It has been shown that parrot presence is often influenced by food availability (Renton et al. 2015) therefore recording if trees have flowers, fruits or seeds, and quantifying this using established methods for censuring fruiting phenology such as fruit traps and ground raking (Zhang & Wang 1995) may help to further understand *A. lilacina's* movements. Furthermore, the inclusion of additional landscape scale characteristics such as habitat patch size and connectivity, land use cover, distance to roosting sites and known nesting sites, may help to improve the predictive ability of our models.

A number of our results suggest that the protected area is the most suitable habitat for *A. lilacina* in our study. It had a much greater species richness of feeding trees, with 33 compared to just five in the community owned land where previous research suggests that at least at certain times of the year, *A. lilacina* feed predominantly on just two species (Blanco et al. 2016). Furthermore, it had significantly lower evidence of human disturbance and significantly greater evidence of animal biodiversity, compared to the community owned land and the recovering agricultural land. Despite this, it is clear that these areas are all very important for *A. lilacina*. In light of a recent prediction that for parrots worldwide, future agricultural expansion will continue to have negative consequences on their survival (Vergara-Tabares et al. 2020), it is essential to work with local communities to enable them to preserve habitats that are suitable for parrots, within their community-owned lands.

Conclusion and further research

The habitat assessments presented in Chapter Four show that *Amazona lilacina* displays preferences for certain dry forest characteristics, and this varies depending on the time of day or behaviour. They also suggest that they use only a few tree species for nesting and roosting, and that three of the four roosting species are mangroves which are globally threatened (Hamilton & Casey 2016). The distribution of this species is likely to be dependent on the availability of suitable habitat for communal roosting, and it is possible that communal roosting can help to modify foraging behaviour. These Amazons appear loyal to roost sites, however the function of their communal roosting behaviour is unclear (Berg and Angel 2006, Kunz 1996, Ridgely and Greenfield 2001) - Chapter Five aims to study this roosting behaviour in more detail.

Chapter Five: Using roost surveys to assess population dynamics of *Amazona lilacina*

Abstract

Amazon parrots often utilise traditional communal roost sites, which they are loyal to for many years and return to each night. Surveys of such roosts, recording the total number and composition in which birds arrive or depart, can be used to estimate population size and reproductive parameters such as the number of nest attempts or fledged chicks. We conducted roost surveys for 36 consecutive months between 2016 and 2019. Parrots were recorded in the morning when departing the mangrove roost site, and in the afternoon as they returned after a day foraging in the nearby dry forest. We compared the results of morning and afternoon surveys using paired t-tests and estimate the total roost size based on maximum counts. We estimate the population at 149 birds, which is the mean of the highest five counts ranging from 135 to 173 individuals. We found significant differences between the overall morning and afternoon counts. We used generalised mixed linear models, to assess the significance of month on roost size and composition i.e. if birds were flying as a pair, single or triplet. The breeding season for this species is from November – March. We found that the roost was significantly larger in January, February and March and significantly smaller in November, December and April. This contradicts what is generally understood for amazon parrot roosts, and what was seen at this roost twenty years ago, where the peak roost size is during the non-breeding season. We found that a significantly greater proportion of the roost was made up of single birds in November, and of birds flying as triplets in May. However, because of the overriding unusual trend seen in overall roost size, it is difficult to further analyse roost

composition to estimate the population reproductive output. This seasonal fluctuation in roost size, which contradicts what was reported here twenty years ago raises concern over the stability and security of this roost site, as does the evidence of nest poaching during the study period.

Introduction

Amazon parrots are usually inconspicuous during the daytime when feeding, and many species are only visible at dawn and dusk when gathering or flying to and from communal roosting grounds (Snyder et al. 2000; Ridgely & Greenfield 2001a; Enkerlin-Hoeflich et al. 2006). The study of such roosts has therefore long been a focus of researchers who are eager to understand more about the population ecology of these secretive birds. Roost surveying is relatively straightforward and cost effective, and can be as informative as other methods such as line transects, point transects and mark-resighting (Gnam & Burchsted 1991; Martuscelli 1995; Casagrande & Beissinger 1997; Cougill & Marsden 2004). Previous studies have used roost surveys to estimate population size and to evaluate long-term population trends (Gnam & Burchsted 1991; Martuscelli 1995; Casagrande & Beissinger 1997; Wermundsen 1998; Cougill & Marsden 2004; Matuzak & Brightsmith 2007). In addition, assessment of seasonal fluctuation of roost size has been used to help understand aspects of population ecology, such as the proportion of a population that is breeding, the number of nest attempts, the number of fledged chicks, and the relationship between roosting strategies and foraging (Berg & Angel 2006; Matuzak & Brightsmith 2007; De Moura et al. 2010; Seixas & Mourão 2018; Wright et al. 2019).

Interpretation of roost survey results requires a prior knowledge of Amazon parrot behaviour. Firstly, that birds are loyal to roost sites over the long-term, often using them every evening “as long as the trees are standing” (Martuscelli 1995) therefore lack of attendance at a roost site is likely due to a factors such as adverse weather or human disturbance (Snyder et al. 1987; Cougill & Marsden 2004). Secondly, that their monogamous social systems makes pairs conspicuous in flight and thus study of the composition of groups flying to the roost may indicate social structure (Enkerlin-Hoeflich et al. 2006). Finally, Amazon parrots have seasonal breeding with clearly defined parental roles, and thus attendance at the communal roost and the composition of the communal roost may fluctuate across the year. For example, females are responsible for incubating eggs in the nest cavity and therefore will not return to the communal roost at night during this time (Snyder et al. 1987; Berg & Angel 2006). In many cases roost size may diminish gradually as the breeding season advances when breeders roost in nests, and may sharply increase after young fledge and follow their parents to the roost (Dénes et al. 2018).

There are a number of best practices to consider when developing roost survey protocols (Dénes et al. 2018). The location of neighbouring roost sites may influence the daily variation in roost counts; therefore roosts must be independent, for example it was believed for Red-tailed Amazons, *Amazona brasiliensis,* that there would be relatively low day-to-day movement between two roost sites that were 8 km apart (Cougill & Marsden 2004; Dénes et al. 2018). Prior knowledge of the roost site and local area in question is essential and the location from which to conduct roost surveys must be carefully selected to allow a wide view over the roost site to see birds arriving or leaving, and multiple locations must be trialled in advance (Dénes et al. 2018). In some cases birds may fly in and out of the roost site repeatedly, therefore a method which allows this to be accounted for is necessary – i.e. an imaginary line that birds are counted in and out of when they pass (Cougill & Marsden 2004). If it is not possible to conduct counts every day of the year, the counting regime must be carefully considered. It has been found that a random selection of dates over a month (five to ten) or counts over successive days performed better at estimating roost size than returning periodically, i.e. retuning every four days within the same period (Cougill & Marsden 2004). A number of studies have found that counts conducted in the morning, when birds are passing closer to the observer, are more accurate than evening counts (Cougill & Marsden 2004; Berg & Angel 2006; Matuzak & Brightsmith 2007). This, however, cannot be assumed for all species and roost sites, thus morning and evening surveys must be conducted at least in the first instance to assess which is more reliable.

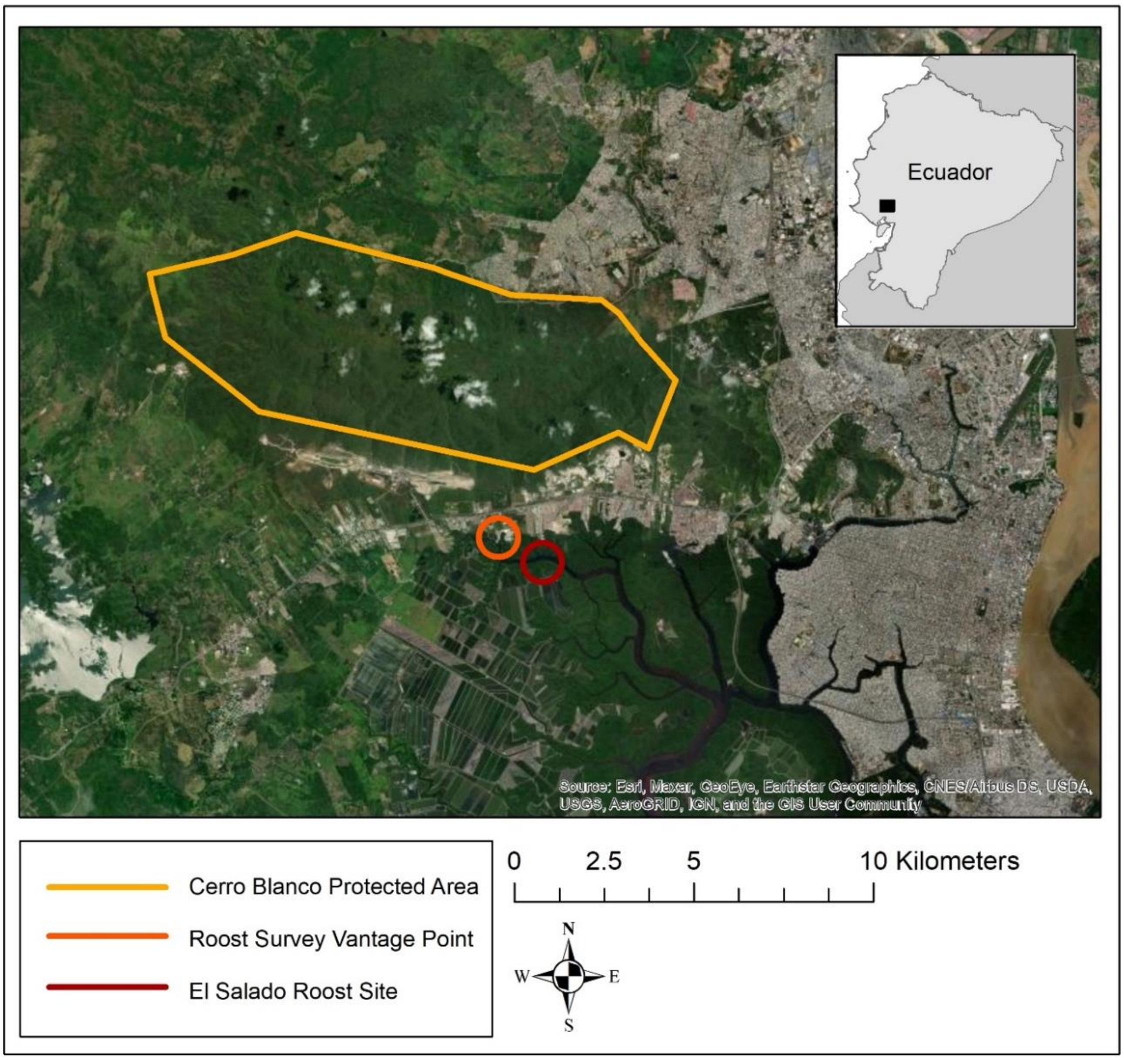
Even when a robust protocol is developed, imperfect detection is a factor that must be considered (Dénes et al. 2018). Observer bias in detection can be minimised by ensuring that the same researcher conducts all counts, and sampling effort can be controlled for by ensuring the observer is present for the same amount of time each day, which encompasses all plausible flight times of the birds. For this study, we developed a roost survey protocol following best practice, and conducted roost counts at a long-standing independent roost of the Critically Endangered *Amazona lilacina*, for three years with the following specific aims:

i. estimate the size of the subpopulation from the maximum number of birds observed during roost surveys;
ii. determine if there is a difference between the number of birds observed during morning and afternoon surveys, and which is likely to give a more reliable representation of roost size and seasonable fluctuation;
iii. assess whether roost surveys can be used as an indicator of population reproductive health.

Methods

Study area

For this study, we focused on the subpopulation of *Amazona lilacina* which forages and nests in the dry forests of the Cordillera de Chongón, in particular the Cerro Blanco Protected Forest, and roosts in the mangrove islands of the El Salado Reserve (Berg & Angel 2006). Birds travel between the two areas at dawn and dusk, flying over the fishing town of Puerto Hondo, a behaviour that has been observed by local communities since at least the early 1990s (Figure 1). It is understood that all *A. lilacina* individuals from the local area return to this roost site, which is believed to be independent (> 40 km away) from other roosts (Berg & Angel 2006; Biddle et al. 2020).



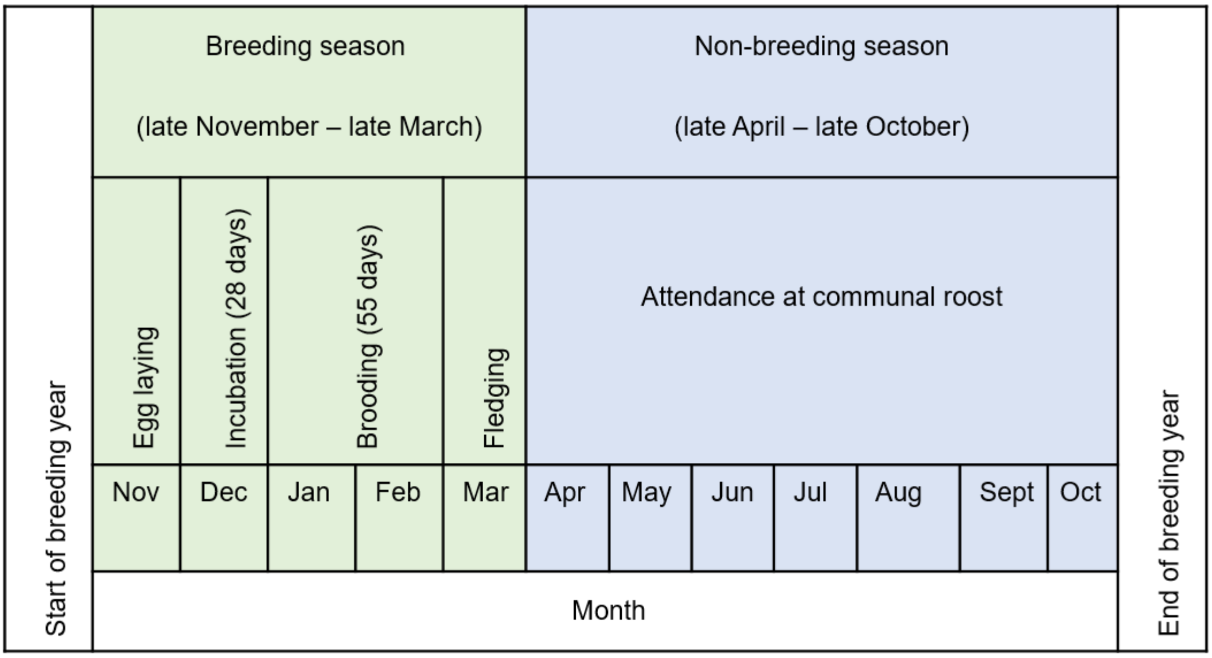
Source: Esri, Maxar, GeoEye, Earthstar Geographics, CNES/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community

Figure 1: Map showing the Cerro Blanco Protected Area that is used by *Amazona lilacina* for foraging and nesting, the El Salado Mangrove Reserve where parrots roost overnight, and the vantage point from which roost surveys were conducted. The large city of Guayaquil is shown to the east of the roost site, and the rectangular clearings of mangrove to the south of the roost are shrimp farms.

Reproductive timeline

Parental responsibilities of *A. lilacina* during the breeding season may affect the attendance of birds at the roost site, as they may choose to roost overnight in nest cavities in the dry forest, rather than flying to the mangrove (Berg & Angel 2006). The breeding season is reported as January – March (Juniper & Parr 1998). From closely monitored active nests within Cerro Blanco Forest it appears that adults begin nesting from November and produce one or two chicks that fledge between mid-February and late March the following year (Kunz 1996; Berg and Angel 2006, Fundación Pro Bosque reports and R. Biddle *pers. obs.* 2017 - 2020). For this reason, we define the "breeding year" in this study, from November to October. The fledging date is also supported by studies of *A. autumnalis* (*A. lilacina's* closest relative) where 24 nests were monitored in north-eastern Mexico. The mean length of incubation was 28 days, followed by 55 days of brooding before chicks fledged between the 19th of March and the 22nd of April (Enkerlin-Hoeflich et al. 2006). Prior to egg laying, the pair may roost in the nest cavity overnight rather than returning to the communal roost, then once eggs have been laid, the female is expected to roost in the nest cavity, whilst the male may return to the communal roost alone (Berg and Angel 2006). In the early stages of brooding one or both of the parents will roost in the nest cavity or nearby, but as chicks get larger there is less room in the cavity for roosting adults (Enkerlin-Hoeflich et al.

2006). After fledging, chicks are likely to spend one / two months around the nest cavity before making the journey to the mangrove roost site with their parents (Figure 2).



<table>
<tr><td rowspan="4">Start of breeding year</td><td colspan="5">Breeding season
(late November – late March)</td><td colspan="7">Non-breeding season
(late April – late October)</td><td rowspan="4">End of breeding year</td></tr>
<tr><td>Egg laying</td><td>Incubation (28 days)</td><td colspan="2">Brooding (55 days)</td><td>Fledging</td><td colspan="7">Attendance at communal roost</td></tr>
<tr><td>Nov</td><td>Dec</td><td>Jan</td><td>Feb</td><td>Mar</td><td>Apr</td><td>May</td><td>Jun</td><td>Jul</td><td>Aug</td><td>Sept</td><td>Oct</td></tr>
<tr><td colspan="12">Month</td></tr>
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Figure 2: Probable reproductive timeline of *Amazona lilacina*, showing parental responsibilities where they may affect parrots' attendance at the communal roost site.

Roost survey protocol

We investigated possible vantage points from which to conduct roost surveys that were situated between the roost site, and the initial destination of the birds - the hills of Cerro Blanco Forest (Figure 1). We decided the observation tower in Puerto Hondo was the best vantage point, due to its proximity to the roost site (~700 m) which made the birds highly visible and audible, and its raised position (~20 m) which allowed observers to look over the mangroves and provided a wide field of view around the roost site. Our initial investigations revealed that the bird's flight was always very determined and purposeful; they would only either fly into the roost in the afternoon or away from in it the morning (other species birds may fly in and out of the roost multiple times in one evening). This meant the risk of double counting birds was minimal. Birds were very vocal during flight therefore relatively easy to identify by their call. To reduce observer bias, all surveys were conducted by Paùl Cun of Fundación Pro Bosque, who attended the vantage point from 05hr30-07hr30 and 17hr30-19hr30 and recorded the number of birds and the composition in which they were flying (Figure 3). Although observers were present for two hours in the morning and evening, most birds depart and arrive at the roost site within 20 minutes, therefore our increased lengths of observations were to enable us to be sure that all birds were counted. Surveys were carried out in all weather conditions, in the morning and evening of the last four days of every month between November 2016 and November 2019.

Figure 3: Photographs showing the views in opposite directions from the observation tower viewpoint; the roost site and the Cerro Blanco Forest. From the viewpoint, the observer recorded the number and composition of birds that were seen flying past; photographs show examples of a pair and triplet group of *Amazona lilacina*.

Data analysis

To assess how accurate observer counting was, we compared the number of birds counted in the PM, with the number counted the following AM. This should represent the same birds arriving in the afternoon, then departing the following morning, and so if the observer is successfully counting all birds, these numbers should be similar. For this we used a paired-t test, and examined the total number of birds counted, for each of these paired examples each month.

Then to assess the overall difference between the results of AM surveys compared to the PM surveys, we used a paired t-test, but with counts paired within the same day, i.e. the AM survey paired with the PM survey conducted later that same day. This was important because a previous study from this roost site showed a significant difference between AM and PM roost counts, and concluded that counts from the AM were likely to be more accurate than those conducted in the PM (Berg & Angel 2006). They suggest this is because AM roost surveys were conducted ~700 m from the roost site, thus if the birds leave the roost at ~10 m above the ground on the same small island they are much closer to the observer, than in the PM when they arrive from a considerable distance (2-7 km) and height (50-500 m). We also used a two-sample test of variance to look for differences within the data from AM compared to from PM counts.

In order to determine whether month had a significant effect on the size or the composition of the roost, we combined the three years of data, and used generalized linear mixed models with the total number of birds, and then the percentage of the roost made up of singles, pairs or triplets as the response. We included month as a fixed effect, and breeding year as a random effect. The family was set to Poisson to account for the distribution of our count data. We defined the reference month (intercept) as August, as this is when there was the least chance of the reproductive timeline or parental responsibilities, affecting the number of birds attending the roost or the roost composition. Models were built in R using the *glm* function of the “lme4” package (Bates et al. 2020). The function *rsquaredGLMM* in the package "MuMIn" (Barton 2020) was used to calculate the variance explained by the fixed effects in each model.

Results

Observed roost size and difference between AM and PM counts

We conducted 300 roost surveys; 90 in year one, 94 in year two and 116 in year three. These showed a mean of 43 (min 2 - max 121), 38 (min 1 – max 135) and 55 (min 0 – max 173) in years one, two and three respectively. Our estimation of population size, based on the five highest counts (range 135 – 173) is 149 individuals. When looking at paired PM and AM counts as a measure of observer accuracy, there is no significant difference in most months, as would be expected. However, in March, May, July and September there is significant difference between the PM and the following AM counts. In four months, the AM count was higher than the PM count, but in general, there were more birds counted in the PM than in the AM (Table 1).

When looking for an overall difference between AM and PM counts by comparing those within the same day, we see a significant difference in the total number of individuals and the number of pairs counted in the AM compared to the PM on the same day (Table 2). A two sample test of variance showed that there is no significant difference between the variance within AM counts compared to PM counts (p = 0.319, F = 0.763, df = 149) however, the boxplots show that when data are combined, the overall variance appears larger (Figure 4). Therefore, to minimise this variance, we follow the conclusions of Berg and Angel 2006 who suggest that AM counts are likely to be more reliable, and use only AM counts for further analysis.

Table 1: The number of birds observed attending the roost each month, averaged across breeding year one, two and three. The difference between the numbers of birds arriving in the PM, compared to departing the following morning as a measure of observer accuracy.

Month	Mean of all counts	Mean of AM counts	Mean of PM counts	Difference between paired counts; PM paired with following AM		
				p	t	df
November	23	25	21	0.251	1.253	7
December	27	28	25	0.175	1.508	7
January	63	64	62	0.259	1.204	9
February	83	82	83	0.821	0.234	8
March	21	64	34	0.011	3.168	9
April	28	26	31	0.731	0.355	9
May	48	38	59	0.028	2.622	9
June	29	31	54	0.128	1.677	9
July	59	46	71	0.030	2.564	9
August	40	34	45	0.184	1.440	9
September	41	33	48	0.042	2.373	9
October	49	40	58	0.115	1.771	8

Table 2: The results of a paired t-test to indicate differences between the number of birds counted during AM surveys compared to PM surveys on the same day.

	p	t	df
Difference between total number of birds counted	0.015	2.44	149
Difference between number of singles counted	0.090	1.70	149
Difference between number of pairs counted	0.006	2.80	149
Difference between number of triplets counted	0.355	0.93	149

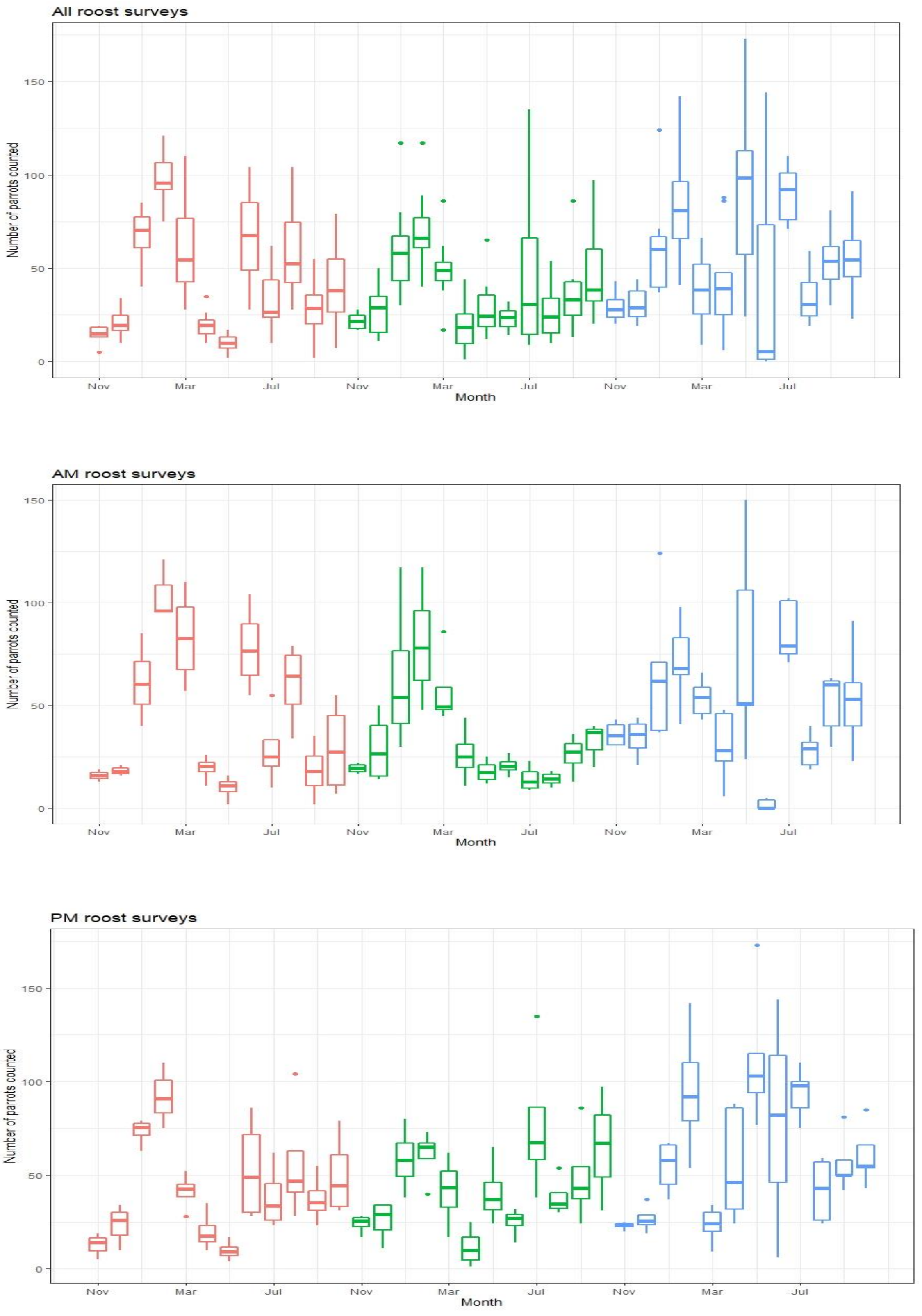


Figure 4: The number of birds observed at the roost site for 36 months (year one in red, year two in green, year three in blue). Results are combined (a), and then split into AM surveys (b) or PM surveys (c).

Fluctuation in roost size and composition

There is a peak in roost size during January, February and March (Figure 5). In all months, the roost consists mainly of paired birds, with fewer birds flying as triplets or singles. To account for the confounding roost size, i.e. the chance of there being more singles in a particular month as a result of an overall larger roost size, we calculated the percentage of the roost that was made up of singles, pairs and triplets, from the observed AM counts each month. Across all months, the majority of the roost is made up of pairs, but there is a peak in the proportion of triplets in May, and a peak in the proportion of singles in November (Figure 6).

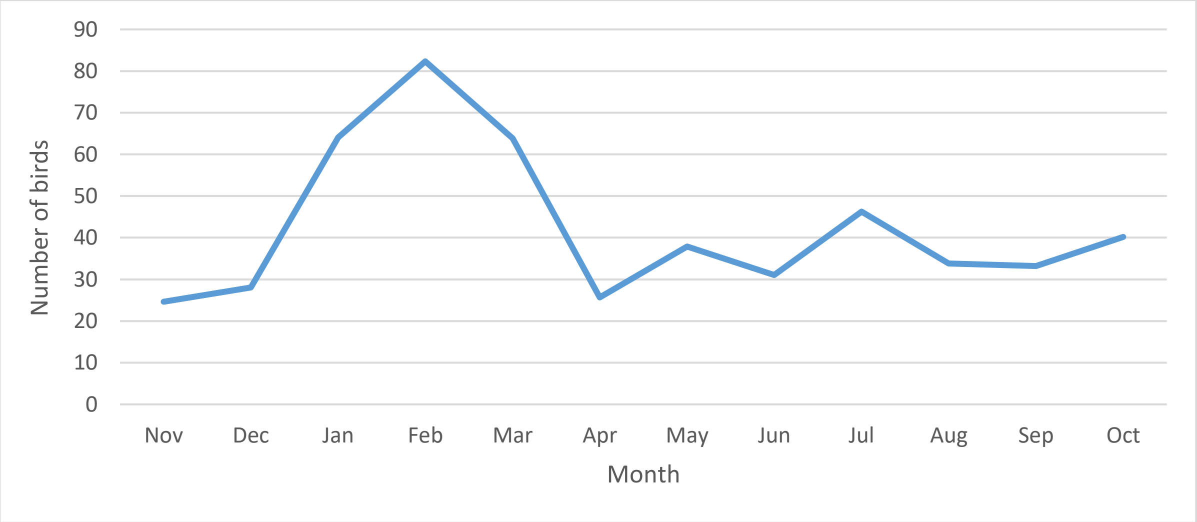


Figure 5: The average number of birds observed departing the roost site in the AM, per month, for three years.

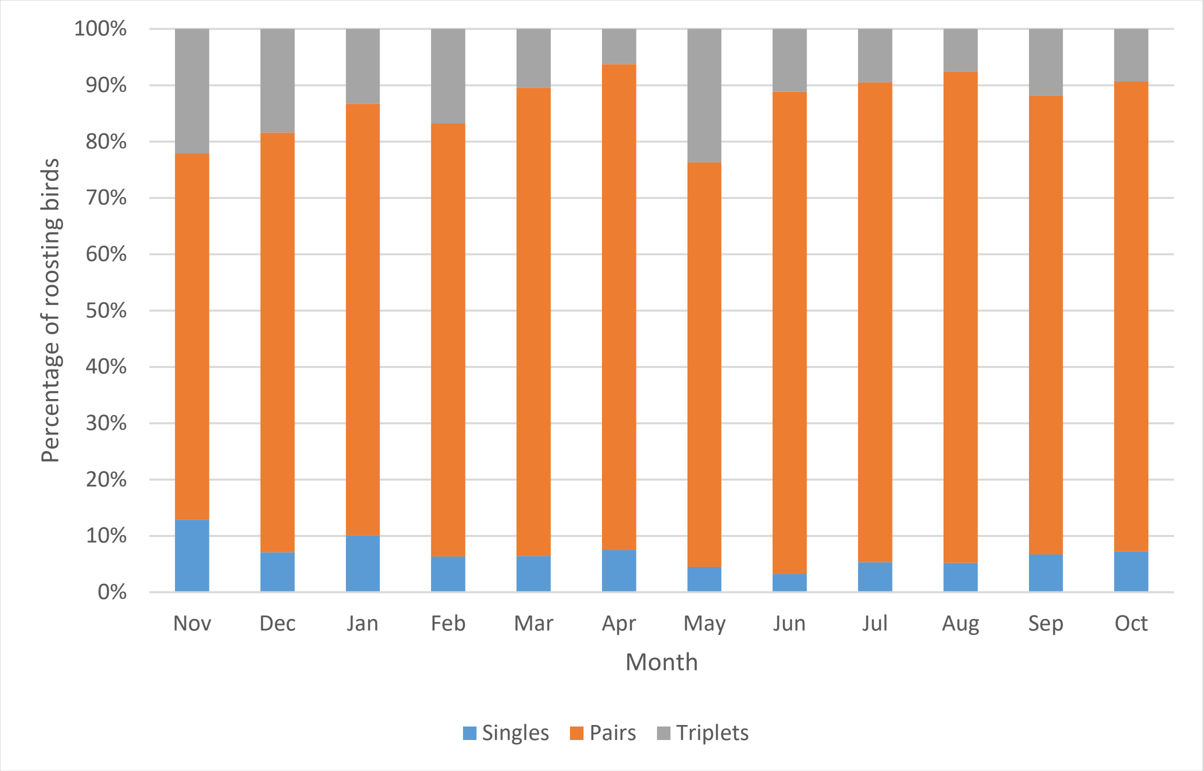


Figure 6: The observed roost composition of birds recorded departing the roost during morning surveys for three years.

The results of our GLMs showed that, in many cases, month had a significant effect on the number and composition of birds attending the roost site. The R^2 values show that month explained 85% of the variance for the total number of birds attending the roost, 84% of variance for the number of pairs attending the roost, 65% of variance in the number of triplets attending the roost, and 46% of variance in the number of singles attending the roost. There are significantly more birds attending the roost in the months of January, February and March, and the overall roost size is larger in May, July and October compared to August. When considering roost composition, single birds make up a significantly greater proportion of the roost in November, January and July and a significantly greater proportion of the roost is made up of triplets in March, April and May (Table 3).

Table 3: The estimated coefficients (β) and significance values from our GLMs looking at monthly total roost size and proportional composition. A positive β suggests that during that month the number of birds / proportion is greater than in August, whilst a negative β suggests it is lower.

Month	Total number of birds		Singles (proportion)		Pairs (proportion)		Triplets (proportion)	
	β	p	β	p	β	p	β	p
August	3.5	<0.001	1.9	<0.001	4.4	<0.001	-0.3	0.027
November	-0.3	<0.001	0.7	<0.001	-0.3	<0.001	-0.1	0.239
December	-0.2	0.012	0.1	0.371	-0.1	0.015	0.3	0.033
January	0.6	<0.001	0.4	0.002	-0.1	0.117	0.3	0.020
February	0.9	<0.001	-0.1	0.694	0.0	0.437	0.2	0.165
March	0.6	<0.001	0.0	0.755	0.0	0.969	0.9	<0.001
April	-0.3	<0.001	0.1	0.352	0.0	0.620	0.5	<0.001
May	0.2	0.008	-0.2	0.251	-0.2	<0.001	0.7	<0.001
June	-0.1	0.215	-0.4	0.015	-0.3	<0.001	-0.1	0.594
July	0.3	<0.001	0.5	<0.001	-0.1	0.022	0.1	0.367
September	-0.1	0.186	0.0	1.000	0.0	0.573	0.0	0.572
October	0.2	0.009	0.2	0.257	0.0	0.932	-0.2	0.087

Discussion

Our estimate for the size of this subpopulation is 149 birds, which is considerably lower than the 214 estimated using the same methods twenty years ago (Berg & Angel 2006). There is monthly variation in attendance of Amazon parrots at this roost site, however the seasonal pattern contradicts with both the typical fluctuation in Amazon parrot roost size (Dénes et al. 2018) and the pattern that was seen at this roost site twenty years ago (Berg & Angel 2006). Normally, more birds are present during the non-breeding season compared to the breeding season; however, our results suggest the opposite for this roost site, which reaches its peak size in January – March, to coincide with the breeding season. This raises concern over possible disruptions to this roost site in the last twenty years. Because of this, it is difficult to estimate the proportion of birds that may be reproductively active.

Results indicate that observer efficiency in counting birds at this site is high, with no difference between the numbers of birds counted departing the roost site in the morning compared to the number that arrived the previous afternoon, in eight months of the year. This high observer efficiency suggests that our population estimate for this roost site - 149 birds - is likely to be fairly reliable. However, during the four months where there was a significant difference between afternoon counts and the following morning counts, there may be external factors such as weather conditions that reduce the chance of observing all the parrots. Previous studies of Amazon parrot roost sites, show factors such as weather conditions which improve the observers' vision, and parrot behaviour such as their height of flight, can impact the observers' counting efficiency (Cougill & Marsden 2004; Berg & Angel 2006; Dénes et al. 2018).

There were significantly more birds attending the roost during the breeding season, January-March. This contradicts the results from a previous study of the same roost site from 1999, which shows a peak roost size in the months of May, June, July and September which all fall outside of the breeding season (Berg & Angel 2006). Our results also contradict those of many other Amazon parrot roost studies that have found the size of communal roosts to be larger during the non-breeding season (Martuscelli 1995; Casagrande & Beissinger 1997a; De Moura et al. 2010; Tossas et al. 2012). This trend is often perceived to be so clear that many studies designed to estimate roost abundance, only conduct surveys during the non-breeding season (Gnam & Burchsted 1991; Cougill & Marsden 2004; Rodríguez-Ferraro & Sanz 2007; Wright et al. 2019). This change in the seasonal pattern of roost size, i.e. more birds during the breeding season, raises questions about the function of this roost and the condition of the local area. It is possible that birds are relying on this roost site during the breeding season, because there is a limitation in dry forest nesting grounds elsewhere, and this roost site is close to the Cerro Blanco Forest where birds can successfully nest. Additionally, it is possible that birds are leaving this roost site during the non-breeding season due to disruption from the expanding city of Guayaquil and local shrimp farms. We must also consider that our reproductive timeline is based on just a handful of observed nests and therefore may not be representative of the whole population, so future resources to help locate and monitor when chicks fledge would be extremely useful.

Although we did see significant differences in the percentage of the roost composed of singles, pairs and triplets, across the months, the overriding pattern of the total roost size, which conflicts with what is normally seen in amazon parrots roost, means that any further analysis of this composition in order to estimate reproductive output of the population is potentially unreliable. The proportion of singles peaked in November, which supports our proposed reproductive timeline and previous work that suggests this peak is due to males returning to the mangrove roost site alone, leaving their female mate to lay or incubate eggs in the dry forest nest cavity (Kunz 1996; Berg & Angel 2006; Enkerlin-Hoeflich et al. 2006). This is also seen in other species e.g. the Puerto Rican Amazon *A. vittata* where in the majority of cases, the female roosts in the cavity after the eggs are laid (Snyder et al. 1987). Although this is not a direct measure of reproductive success, for the Glossy Black Cockatoo *Calyptorhynchus lathami* the number of pairs attempting to nest was the principal determinant of annual productivity (Cameron 2009). The peak in the percentage of the roost made up of triplets in May also fits our reproductive timeline, and previous work which suggests that young Yellow-naped Amazons *A. auropalliata* start to fly with their parents to the communal roost site one to two months after fledging (Matuzak & Brightsmith 2007). Therefore, if *A. lilacina* fledge mainly in late March this peak in triplets in May could represent fledged chicks.

Whilst it has been shown that roost counts offer a robust method of estimating population size (Gnam & Burchsted 1991; Martuscelli 1995; Casagrande & Beissinger 1997; Wermundsen 1998; Cougill & Marsden 2004; Matuzak & Brightsmith 2007), it is important to also note their limitations. Namely, that at different times in the year, birds may choose to roost in smaller more transient roosts, based on factors such the distribution and availability of food (Seixas & Mourão 2018). Therefore, it is essential to conduct surveys throughout the entire year, ideally across a number of years, and to spend time investigating the surrounding areas for small transient roosts. Furthermore, interpretation of seasonal fluctuation should be considered alongside not only the reproductive timeline, but also the phenology of important feeding tree species. Another factor that must be taken into account is that of imperfect detection - resulting from either the observer not successfully counting all birds arriving at / departing the roost site, or the birds not attending the roost site every night (Dénes et al. 2018). Future studies should investigate the use of more advanced statistical analysis methods, such as N-mixture models that allow a detection probability to be calculated and accounted for in estimation of monthly roost size (Royle 2004; Zulian et al. 2020).

The results of this study, in comparison to twenty years ago, suggest that increased habitat degradation in this area is affecting *A. lilacina’s* roosting dynamics (Berg & Angel 2006). Communal roosting appears to be an extremely important behaviour for this species and we suggest that roost monitoring should continue to allow long term population monitoring, but also to further understand roost function. Future work should also look to replicate this study at all remaining roost sites to assess roosting dynamics for the entire global population.

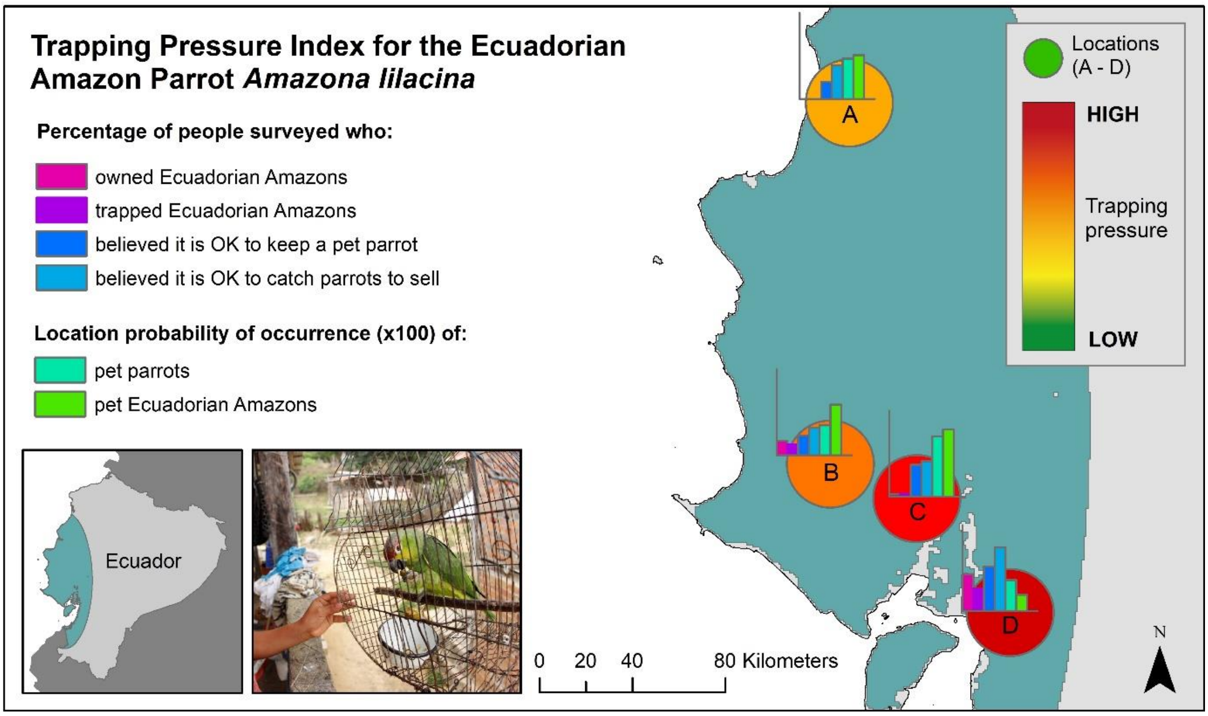
Conclusion and further research

The roost surveys presented in Chapter Five highlighted that the subpopulation of *Amazona lilacina*, attending the El Salado Mangrove roost site near the big city of Guayaquil, has declined in the last 20 years. However, we were unable to estimate information regarding the reproductive output of this population as has been done in previous studies. Due to the species cautious and secretive behaviour at nest sites (Kunz 1996; Berg & Angel 2006; Enkerlin-Hoeflich et al. 2006), between 2016 and 2020 only five nests were located and monitored, using three different cavities in the Cerro Blanco Forest. In 2019, one cavity was destroyed in a storm during which the main trunk snapped at the point of the cavity, trapping the chick inside the rotten cavity on the forest floor. Luckily, this nest was being monitored by Fundación Pro Bosque staff, who were able to excavate the chick the next day and rehabilitate it, allowing it to re-join the wild population. Then, in 2020, another cavity was destroyed when the tree was purposefully cut down and the parrots were trapped and removed prior to fledging.

For parrots worldwide, the threat most closely associated with decreasing population trend is capture for the local pet trade (Berkunsky et al. 2017). For *A. lilacina*, thousands of birds were exported internationally in the early 80s (CITES 1986). This became illegal in 1984 (Ecuadorian National Assembly 2017) and since then there is little information on the number of birds that are kept locally as pets, or indeed the number of birds that are trapped by local rural populations. Furthermore, there is little known about the local attitude towards parrots - in a number of cases parrots are viewed as crop pests (Canavelli et al. 2014) and are persecuted because of this (Trainor 2002). Therefore, the next research priority was to understand local attitudes towards *A. lilacina*, the frequency of behaviours such as trapping and pet keeping, and the possible drivers behind these behaviours.

Chapter Six: Parrot ownership and capture in coastal Ecuador: developing a trapping pressure index

Biddle R, Solis-Ponce I, Jones M, Pilgrim M, Marsden S. 2021. Parrot Ownership and Capture in Coastal Ecuador: Developing a Trapping Pressure Index. Diversity **13:**15.



Abstract

We located rural communities with pet parrots and used these locations to predict the probability of illegal parrot ownership across coastal Ecuador, using variables related to demand for pets, parrot availability, and trapping accessibility. In twelve pet keeping communities we carried out in-depth interviews with 106 people, to quantify ownership, trapping, and interviewees' attitudes towards these behaviours. We combined these data to calculate a trapping pressure index for four key roosting, feeding and nesting sites for the Critically Endangered Ecuadorian Amazon parrot *Amazona lilacina.* We found that 66% of all communities had pet parrots and 31% had pet *A. lilacina*. Our predictive models showed that pet parrot ownership occurs throughout coastal Ecuador, but ownership of *A. lilacina* by rural communities, is more likely to occur within the natural distribution of the species. The number of people per community who had owned *A. lilacina* in the last three years varied from 0% - 50%, as did the number of people who had trapped them from 0% - 26%. We interviewed ten people who had captured the species in the last three years, who reported motives of either to sell or keep birds as pets. Attitudes towards pet keeping and trapping differed among the twelve communities: 20% - 52% believed it was acceptable to keep pet parrots, and for 32% - 74% it was acceptable to catch parrots to sell. This said, most people believed that wild parrots were important for nature and that local people had a responsibility to protect them. We conclude that trapping pressure is greatest in the southern part of *A. lilacina*'s range, and urgent conservation measures such as nest and roost protection, and local community engagement are needed.

Introduction

Parrots (Psittaciformes) are one of the most endangered and rapidly declining birds groups, with 28% of their species classified as threatened (Olah et al. 2016). Globally, over a third of parrot species are caught to fulfil the demand of the international wildlife trade (Wright et al. 2001; Olah et al. 2016; Dahlin et al. 2018). In the Neotropics, over half of the studied parrot populations are in decline (Berkunsky et al. 2017), and one reason for this is the high demand for the pet trade (Bush et al. 2014). Neotropical species are particularly favoured as pets (Wright et al. 2001; Sinovas & Price 2015) and it is suggested that trapping is a stronger threat to their conservation than habitat loss (Clarke & de By 2013). Amazon parrots and macaws are preferred due to their attractiveness and ability to mimic the human voice (Tella & Hiraldo 2014); this is illustrated in Costa Rica where nearly 20% of households have a pet parrot and half of these are *Amazona* species (Drews 2001). Consequently, the rate at which Amazon parrots and macaws are trapped is much higher than expected considering their availability in the wild (Romero-Vidal et al. 2020).

Trapping risk is highest where parrots are abundant in the wild, where demand is high and where parrots are relatively easy to catch and sell (Pires & Clarke 2012); therefore, trapping pressure may differ across a species' range and also between species. Additional factors found to drive hunting and trapping include overlap with human population (Pires & Clarke 2012; Harrison et al. 2016) and proximity to infrastructure or towns (Benítez-López et al. 2017). Attitudes and subjective norms are also factors that influence decision making (St John et al. 2010) and are therefore likely to affect the level of pet keeping and capture in different areas. In Ecuador, wild bird keeping is illegal (Assembly 2017), and whilst ownership appears to be declining in major cities (Ortiz-von Halle 2018) demand is still high in rural areas, where over half of coastal communities still keep pet parrots (Biddle et al. 2020). The most frequently reported confiscated bird species in the country are those with wild distributions exclusive to this coastal region (Athanas & Greenfield 2016; Ortiz-von Halle 2018). Also, this region is one of the most densely populated and impoverished (Mideros M. 2012) parts of Ecuador, the habitats here have been drastically reduced (Dodson & Gentry 1991) and are greatly underrepresented in the country's national protected areas system (Cuesta et al. 2017).

The Critically Endangered *Amazona lilacina,* a species recently split from the *A. autumnalis* group, is found exclusively within the coastal region of Ecuador (BirdLife International 2018b). CITES reported thousands of individuals of this species being trapped and exported in the early 1980s (CITES 1986) and although frequency of trapping is likely to have reduced significantly in recent years, there are still multiple reports of capture and pet keeping within rural communities (Biddle et al. 2020). An average of 392 wild-caught parrots, including 30 *A. autumnalis* were confiscated annually in Ecuador between 2003 and 2016 (Ortiz-von Halle 2018). Although some of these may be older birds, and they may be either *A. lilacina* or *A. a. salvini,* this figure suggests that some level of trapping is still occurring to fulfil the demand for pets. The goal of this study was to understand the risk of trapping in rural communities and formulate a strategy for conservation support. Specific objectives were:

i. conduct surveys across coastal Ecuador to locate communities with pet parrots;
ii. use the locations of known pet parrots, to predict the distribution of pets throughout the species range, using variables related to parrot availability, opportunity and demand;
iii. interview local people from communities where pet parrots are present, to quantify the level of parrot ownership, trapping and the attitudes towards these behaviours;
iv. develop a trapping pressure index based on model predictions, locally reported incidence and attitudes towards parrot capture and ownership, to highlight areas of possible increased extinction risk.

Methods

Surveys to locate communities with pet parrots

In order to locate rural communities with pet parrots, we conducted surveys between January and July 2017. The study area encompassed the extent of occurrence of the *A. lilacina* and communities close (<10 km) to forest patches, where wild parrots may occur were selected. Participants were asked to confirm if they knew of pet parrots in their community, and if possible to identify the species. Prior verbal consent was obtained from each participant and full ethical approval of survey content and methods was gained from The North of England Zoological Ethical Review Committee. We aimed to survey at least four households per community, however some communities were made up of just a few houses, so this was not always possible. We recorded the geographic coordinates of communities with all pets, pet parrots and pet *A. lilacina* and calculated how many communities each species was recorded in. We used IUCN Red List range maps provided by BirdLife International (BirdLife International 2019) in order to determine if species were native to the study area. Range maps are frequently updated so we report the year of update for each range map in the results. ArcGIS (version 10.8.1) (Environmental Systems Research Institute 2020) was used, clipping the distribution shape files, to calculate the size of each species range within our study area.

It is illegal to keep native bird species as pets in Ecuador (Ecuadorian National Assembly 2017), yet in our experience, people speak openly about their parrots and are proud to show them off. However, it was important that participants did not feel threatened or that we were collecting information to inform the authorities. Therefore, surveys were conducted by a local Ecuadorian researcher, in Spanish, with only the researcher and interviewee present and it was made clear that all information given was anonymous, and only to be used for scientific research.

Distribution models to predict parrot ownership

From our surveys, we created two groups of geographic coordinates to represent 1) communities with pet parrots, and 2) communities with pet *A. lilacina*. The MaxEnt package in R (version 4.0.3) (Phillips et al. 2006; R Core Team 2020) was used to build distribution models based on these coordinates combined with random background points within 30 km buffers of community locations, to predict the distribution of pet parrots, and the distribution of pet *A. lilacina*. Variables were extracted to match each corresponding location and were chosen due to their influence over parrot ownership and trapping (Pires & Clarke 2012): opportunity (presence of parrots and their desirability); demand (presence of people and the infrastructure for trade); and accessibility (into the forest).

For each location, we calculated a "species value" to represent parrot trapping opportunity. For the pet model this was calculated based on the presence of wild parrot species at that location using species range maps (BirdLife International 2019), combined with the frequency of the species being reported as a pet; 0.1 was allocated for each species present in that area, and an additional 0.1 was added if that species was reported in a single community, 0.2 if in two communities etc. This value used just for comparative purposes within the study and we gave equal weighting to wild species presence and popularity in captivity as we had no evidence that either was more important than the other. For the pet *A. lilacina* model, this value was replaced with the predicted occupancy area from our distribution models created using observations of the wild population (Biddle et al. 2020). For both models, we also used the estimated human population (Sorichetta et al. 2015), the Euclidean distance to the nearest town and nearest road calculated in ArcGIS using OpenStreetMap (OpenStreetMap Foundation 2019) data, and the mean annual Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product over 2010-2015 as a proxy of vegetation cover.

For each group of points, spatial autocorrelation was controlled for by limiting them to one per 1 km using the R package spThin (Aiello-Lammens et al. 2015). Predictors were checked for pairwise correlation across random points within the study area, using pair plot for collinearity (Zuur et al. 2009). Model evaluation was performed with five-fold cross validation and the mean AUC +/- SD are presented to demonstrate the predictive ability. An AUC of 0.7 means there is an 70% chance that the fitted model will be able to correctly distinguish between presence and absence (Wisz et al. 2008). All data were included in the final models. We present the permutation importance (%) of variables with a high value indicating that the final model depends heavily on that variable (Phillips et al. 2006).

Interviews to quantify parrot ownership, trapping and attitudes

We selected twelve communities where pet parrots were present to conduct interviews with community members about their experiences and attitudes towards parrot ownership and capture. These communities ranged in size from 50 to 300 people. The reason these sites were chosen was because our focus was on understanding risk to *A. lilacina*, so the selected communities fell within the species extent of occurrence and were <15 km away from key roosting, nesting and feeding grounds (Biddle et al. 2020). These twelve communities were grouped into four clusters (Figure 1). We interviewed at least six participants from different households in each community. Participants were outdoor workers (i.e. agriculturalists, fishers and crab fishers) selected for their familiarity with parrots in their local area. The same methods regarding informed consent and data anonymity as described above were followed. Due to low literacy levels amongst participants, all questions were read out aloud and the answer provided was recorded by the researcher. Age and gender of each participant was recorded. The interview consisted of eight questions and seven attitude statements arranged on a five point symmetric Likert scale (Table 1). The likert package (Jason Bryer 2016) in R (version 4.0.3) (R Core Team 2020) was used to visualise attitude statements. Responses were grouped into positive, neutral or negative and a non-parametric test (Kruskal-Wallis) was used to determine significant differences in responses between the four community clusters.

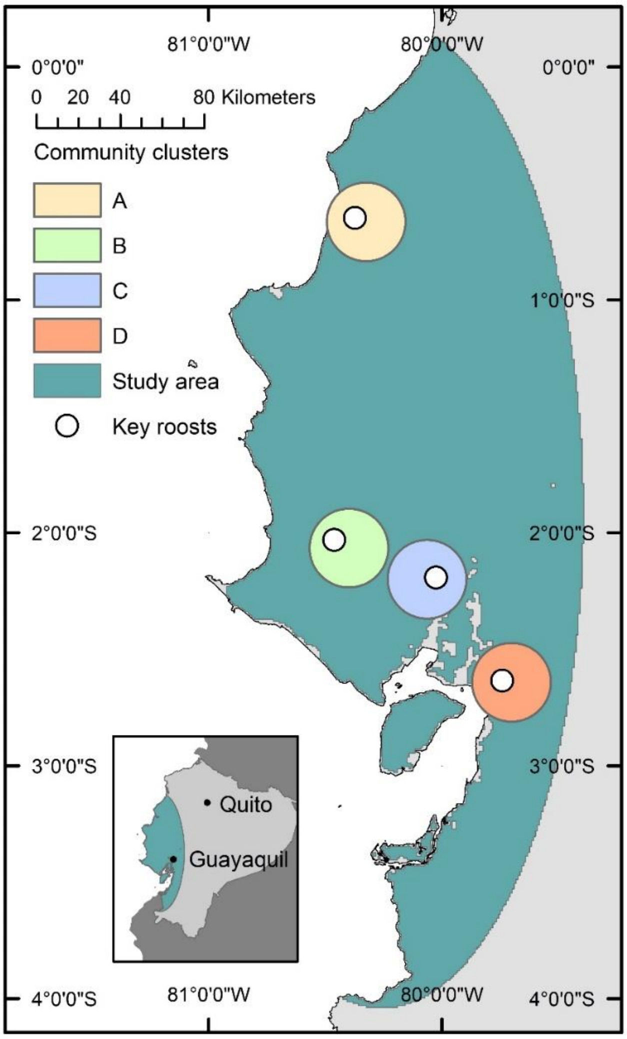


Figure 1: Interviews about parrot ownership and capture were conducted in twelve communities, grouped into four clusters (A, B, C, and D) near key *A. lilacina* roost sites. Each cluster contains three communities <10 km apart.

Figure 2: Examples of pet *A. lilacina* in four rural communities in coastal Ecuador. Parrots were kept in a variety of situations; indoors or outdoors, caged or with clipped wings. In some cases pet parrots that were housed in gardens were not initially considered to be captive by the owner, but for the purposes of this study any parrot living in the locality of people was classed as a pet.

Table 2: The number of communities in which each of the 19 species was reported as a pet and the range size within the study area according to BirdLife International (BirdLife International 2019).

Parrot species (year of update)	Range within study area SA (km²)	Number of communities reporting the species (out of 65)
Lilacine or Ecuadorian Amazon *Amazona lilacina* (2018)	38,860	20
Grey-cheeked Parakeet *Brotogeris pyrrhoptera* (2014)	8,645	20
Red-masked Parakeet *Psittacara erythrogenys* (2007)	54,327	17
Pacific Parrotlet *Forpus coelestis* (2017)	55,300	8
Red-lored Amazon *Amazona autumnalis* (2017)	5,583	6
Blue-headed Parrot *Pionus menstruus* (2013)	27,943	2
Southern Mealy Amazon *Amazona farinosa* (2013)	8,612	1
Bronze-winged Parrot *Pionus chalcopterus* (2014)	46,508	1
White-winged Parakeet *Brotogeris versicolurus* (2018)	549	1
Chestnut-fronted Macaw *Ara severus* (2014)	49,329	0
Blue-fronted Parrotlet *Touit dilectissimus* (2014)	13,470	0
White-capped Parrot *Pionus seniloides* (2012)	1,482	0
Rose-faced Parrot *Pyrilia pulchra* (2002)	12,828	0
Great Green Macaw *Ara ambiguus* (2014)	3,899	0
Red-faced Parrot *Hapalopsittaca pyrrhops* (2000)	49	0
Cordilleran Parakeet *Psittacara frontatus* (2014)	1,347	0
Barred Parakeet *Bolborhynchus lineola* (2014)	2,183	0
Red-billed Parrot *Pionus sordidus* (2014)	1,565	0
El Oro Parakeet *Pyrrhura orcesi* (1999)	615	0
Kept by communities but non-native		
Orange-winged Amazon *Amazona amazonica*	NA	1
Yellow-crowned Amazon *Amazona ochrocephala*	NA	5

Results

Predicted distribution of pet parrots

The locations of the 43 communities with pet parrots and the 20 communities with pet *A. lilacina*, were reduced to 42 and 19 after limiting each group of locations to one per 1 km. A total of 3,803 background points were randomly allocated. The mean AUC of resulting models was 0.69 ± 0.06 (sd) for pet parrots and 0.62 ± 0.20 (sd) for pet *A. lilacina*. The most important variables predicting the presence of pet parrots were distance to nearest road (permutation importance, PI = 40%) and distance to nearest town (PI = 28%); the key factors for the presence of pet *A. lilacina* were the mean annual NDVI (PI = 33%) and species value, representing the native distribution (PI = 27%) (Table 3). Predictions show that pet parrots are likely to be widespread throughout the study area, whereas pet *A. lilacina* seem to be more likely within the species range. Both models show high probability of occurrence of pets to the west of Guayaquil and out towards the coast (Figure 3).

Table 3: Permutation importance values for variables used to create models predicting the distribution of pet parrots and pet *A. lilacina* in coastal Ecuador.

Variable	Permutation importance (%)	
	Pet parrot model	Pet Lilacine model
Mean annual NDVI	11	33
Distance to road	40	1
Human population density	18	23
Distance to town	28	16
Species value	3	27

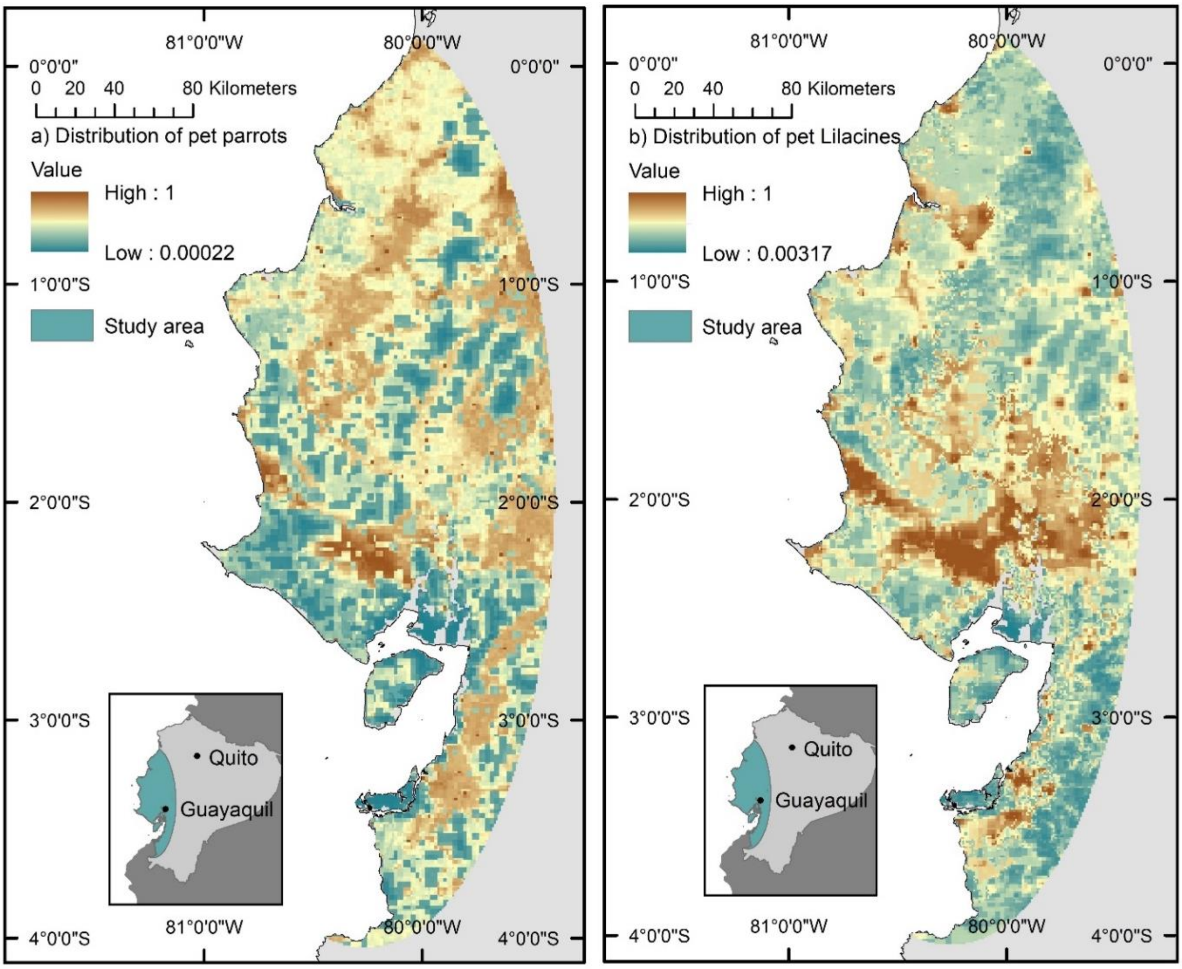


Figure 3: Model predictions showing the distribution of pet parrots (a) and pet Lilacines *A. lilacina* (b).

Incidence of parrot ownership and trapping

Within twelve selected communities where pet parrots occur, 106 (96 men / 10 women) participants (min 6, max 13, mean 8.8 per community) took part in interviews. All participants worked outdoors as farmers (57), fishers (25), crab fishers (18), beekeepers (3) or wildlife guides (3). Participants were familiar with *A. lilacina,* the majority seeing them daily (68%), weekly (19%), or monthly (8%), with the remaining 5% just a few times per year. Of all participants, 66% (70) had owned a pet parrot either previously or currently and 36% (38) a pet *A. lilacina*. The majority (74%, 28) of *A. lilacina* pets had been caught by the owner themselves, with the remainder received as gifts (16%, 6), bought (2%, 6), or found (2%, 6). In the last three years, 15 people have owned a total of 24 *A. lilacina*. 34 people (32%) confirmed that they had previously captured *A. lilacina*, the majority (76%, 26) to keep as a pet themselves, the others to sell (9%, 26) or for undisclosed reasons (15%, 5). Pet ownership and trapping varied between community clusters, with the highest rates of historic and current ownership and trapping of *A. lilacina* occurring in the crab fishing communities (D) in the southern part of the range (Table 4).

Table 4: The number, age and occupations of people interviewed from each community cluster and the number who reported owning parrots or catching parrots, either previously or in the last three years.

Community cluster	n	Mean age (years)	Occupation: farmer (F), fisher (Fi), crab fisher (CF), other (O)				In life time:			In the last three years:	
			F	Fi	CF	O	Owned parrot	Owned *A. lilacina*	Caught *A. lilacina*	Owned *A. lilacina*	Caught *A. lilacina*
A	31	53	8	19	1	3	23	11	9	0	0
B	23	48	23	0	0	0	11	8	8	4	3
C	29	53	26	0	0	3	18	4	4	1	1
D	23	46	0	6	17	0	18	15	13	10	6
Total	106	50	57	25	18	6	70	38	34	15	10

In the last three years, ten interviewees reported that they had caught *A. lilacina*, with at least 16 birds among them, to either keep the bird as a pet (7), to sell it (1), or for an undisclosed reason (2). All had either no or primary level schooling, and were men 23 - 72 years old. They reported seeing wild *A. lilacina* daily (9) or weekly (1), and all but one believed the wild population was stable or increasing. In cases where the capture location was given, this always corresponded to the person's occupation, i.e. farmers reported catching parrots in the forest, fishers and crab fishers reported trapping parrots in mangroves (Table 5). Seven of the ten people who had caught *A. lilacina* in the last three years, reported that multiple other people within their community also catch *A. lilacina*, and all ten know of multiple pet *A. lilacina* in their community (mean 5.2 *A. lilacina*).

Table 5: The age, gender, schooling, and occupation of all interviewees who reported catching *A. lilacina* in the last three years. We report the trapping location, reason for capture and how many were caught.

Community cluster	Age (years)	Gender	Level of schooling	Occupation	Location of capture	Reason for capture	Number of *A. lilacina* caught in last three years
B	41	Male	Primary	Farmer	Dry forest	Pet	1
	23	Male	Primary	Farmer	Dry forest	Pet	1
	72	Male	Primary	Farmer	Dry forest	Pet	1
C	68	Male	None	Farmer	Undisclosed	Undisclosed	1
D	32	Male	Primary	Crab fisher	Mangrove	Pet	2
	54	Male	Primary	Crab fisher	Mangrove	Pet	1
	40	Male	Primary	Crab fisher	Mangrove	Pet	1
	47	Male	Primary	Crab fisher	Undisclosed	Undisclosed	Unknown
	51	Male	Primary	Fisher	Mangrove	Pet	1
	67	Male	None	Fisher	Mangrove	Sell	7

Attitudes towards parrot ownership and trapping

Across all communities, responses to attitude statements show a strong feeling that wild parrots are important for nature and participants indicated that local people have a responsibility to protect the environment. This is mirrored by a strong feeling of discomfort with outsiders coming to catch parrots and with parrots disappearing from their area. On the contrary, 46% of all participants believe it is OK to catch wild parrots to sell and 32% that it is acceptable to keep a pet parrot. Furthermore, 17% of people did not believe that catching wild parrots could make them become extinct in the local area (Figure 4). There were no significant differences between communities in the distribution of positive, neutral and negative responses to all attitude statements apart from one: “I think it is OK to keep a parrot as a pet”. For this there was a significant difference between mean responses of the community groups (H = 6.613, p = 0.022), with 52% of community cluster D believing this is acceptable, whilst just 20% of cluster A believing so.

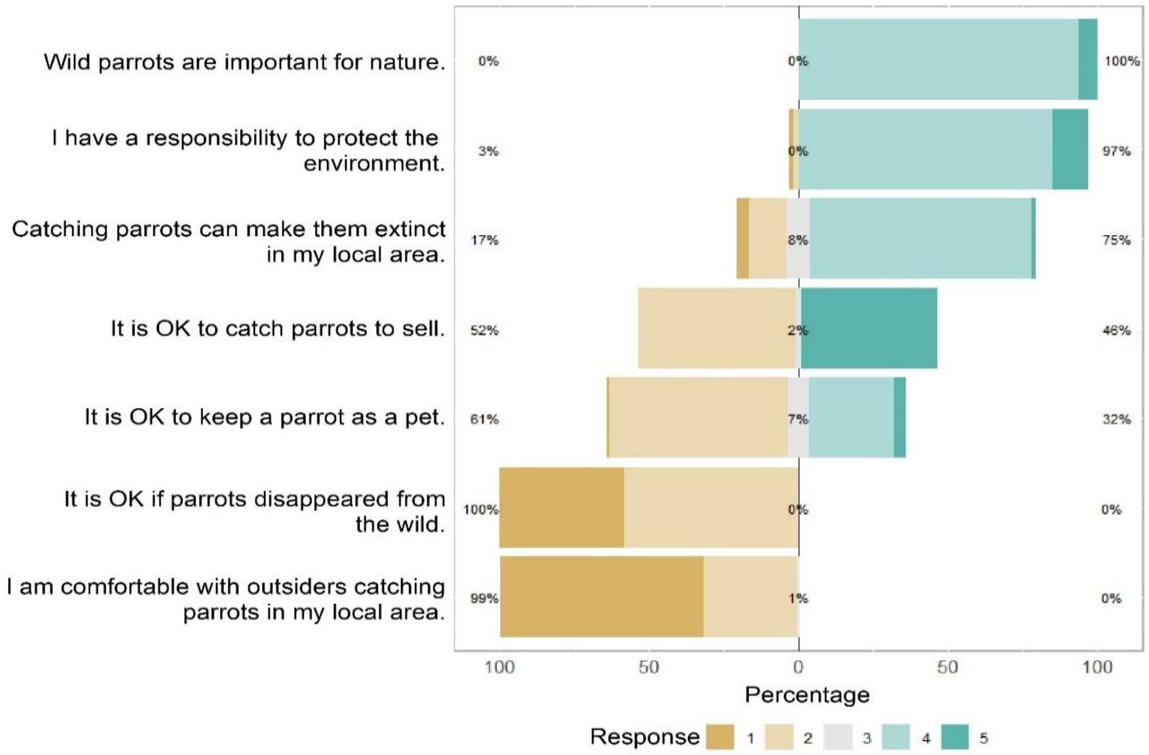


Figure 4: Responses to attitude statements are reported on a five point Likert scale (1 = strongly disagree, 2 = disagree, 3 = neutral, 4 = agree, 5 = strongly agree).

Trapping pressure index

When combining our results into a trapping pressure index we see variation between the four community clusters, with higher values suggesting a higher risk to the wild *A. lilacina* population in that area (Table 6). Wild *A. lilacina* occurring around community cluster D are at greatest risk, due to the high level of reported parrot ownership and capture, and a strong local attitude that this is acceptable. Those occurring around community cluster C are also at high risk, as model predictions here show a high probability of pet *A. lilacina* occurrence (0.78), which suggests a high probability of trapping as pet owners mostly report catching their pet themselves. The wild population occurring around community cluster A appears to be at the lowest risk from trapping, as there were no reports here of current *A. lilacina* ownership or capture, however this is the smallest remaining subpopulation of the species within its range, which could also explain the low prevalence of pets. When considered against participants' responses to their perceived status of the wild population locally, we see more negative responses from the southern community clusters, with the most frequent response in cluster C being 'decreasing' (76%), in cluster D 'stable' (39%), whilst 'increasing' in cluster B (83%) and A (42%).

Table 6: Trapping pressure index for each community cluster, calculated by adding together factors of predicted or reported level of pet ownership and trapping, and attitudes towards pet keeping, capture and trade. Predicted probabilities were converted into percentages for this calculation.

Trapping pressure factor	A	B	C	D
Mean probability of predicted parrot ownership (0-1)	0.47	0.35	0.70	0.36
Mean probability of predicted Lilacine ownership (0-1)	0.51	0.59	0.78	0.19
Percentage of participants who owned pet Lilacines in the last three years	0	17	3	43
Percentage of participants who caught Lilacines in last three years	0	13	3	26
Percentage of participants believing it is OK to keep a pet parrot	20	23	37	52
Percentage of participants believing it is OK to catch wild parrots to sell	39	32	41	74
Overall trapping pressure index (rank)	157 (4)	179 (3)	232 (2)	250 (1)

Discussion

This study found that 66% of rural coastal communities in Ecuador have pet parrots and 31% have pet *A. lilacina.* Within these communities, 66% of people had owed a pet parrot during their lifetime, and 14% currently owned *A. lilacina.* This is similar to Costa Rica, where 18% of households owned a pet parrot in 2001 (Drews 2001). Our current ownership questions focused on just one species, so we expect the level of current ownership of all parrot species to be much higher and similar to Colombia where 58% of all people had pet parrots (Romero-Vidal et al. 2020)[10]. Current ownership and reports of *A. lilacina* trapping in the last three years varied between communities, with 0% to 50% and 0% to 26%, respectively. Ten participants confirmed that they had taken *A. lilacina* from the wild in the last three years to keep birds as pets, or to sell them, so we suggest that similarly to yellow-shouldered amazon *A. barbadensis* harvesting in Venezuela, there are at least two categories of trappers;

“poacher-keepers” and “poacher-sellers” (Sánchez-Mercado et al. 2020), with only the latter having the contacts and logistics to sell birds. People in occupations with an established and frequent trade link, for example fishers or crab fishers, may have more opportunity to transport trapped birds to other towns to sell. We also found that capture location corresponded to the occupation of the poacher, which may explain the variation between rural communities as occupation depends heavily on location i.e. in-land or coastal. Our trapping pressure index identified that the southern distribution of the species is likely to be at greatest risk, fitting with earlier work showing a vast population decline in this area (Biddle et al. 2020) and providing further evidence that this area should be prioritised for conservation support.

Understanding whether taking parrots from the wild is opportunistic or selective is important because selective capture can lead to the extinction of species through overharvesting (Romero-Vidal et al. 2020). Our results suggest that trapping is selective given the differences in the popularity of species, with some kept in 20 communities and some in none. The two most frequently reported pet parrot species differed greatly in body mass, which in general is linked to longevity in captivity [36], suggesting that variation in popularity is not a side effect of survival rates in captivity. Similarly to previous research, there is preference for Amazon parrots, with all three wild occurring species and two non-native species being kept as pets (Tella & Hiraldo 2014). However, parrot ownership and capture, at least within rural communities, may also be opportunistic. Most parrot owners had caught their bird locally, within areas they visit during a normal days’ work and our predictions showed that pet *A. lilacina* were more likely within the species’ wild distribution. Also, parrot species kept as pets had a larger average wild range size than those that are not. This all suggests that ownership and capture are driven in part by parrot availability and accessibility (Pires

& Clarke 2012), but more research including a true measure of wild parrot abundance, and surveys and interviews in larger towns and cities are needed. According to anecdotal reports in some rural communities, orders are placed by outsiders from cities such as Guayaquil or Quito, which fits the typical multi-level chain involving trappers, middlemen and markets described by Pires (Pires 2012) and needs investigation.

Our interviews revealed that *A. lilacina* were trapped both in mangroves, where they roost, and dry forests, where they feed and nest (Berg & Angel 2006; Dupin et al. 2020), suggesting that both adults and chicks are being taken from the wild. Anecdotal reports from communities suggest past events of outsiders casting nets over mangrove islands, to remove an entire roost of *A. lilacina* at a time. Research has shown that the removal of adults from a population can have more drastic consequences on population size and growth rate, than removal of chicks (Valle et al. 2018). In a study of illegal wildlife trade markets in Bolivia, contrary to the idea that most parrots come from nest poaching, 70% of parrots were adults (Pires et al. 2016). Our results also suggest that 60% of Lilacines caught in the last three years were from mangroves, so are likely to be adult or juvenile birds. A number of studies have shown that anti-poaching efforts, in the form of additional human presence, can benefit bird populations (González 2003; Vaughan et al. 2005; Granadeiro et al. 2006) and that recruitment of young people (who may be facilitators in parrot trapping) from the local community to act as nest monitors, can significantly decrease poaching rate (Briceño-Linares et al. 2011). In some cases, nest protection implemented at the correct time of year can have a significant effect (González 2003), but we suggest that year-round protection is needed to safeguard both vulnerable roosting and nesting sites for this species.

The lack of environmental education in Ecuadorian schools is a barrier to reducing parrot ownership and capture (Ortiz-von Halle 2018). The main purpose of any environmental education strategy is to change people's knowledge and attitudes, and ultimately behaviours (Jacobson et al. 2015). Alone, or in combination with other conservation interventions, environmental education projects can result in a decrease in the persecution of parrots and consequently an increase in population size (Sanz & Grajal 1998; Vaughan et al. 2005). Most people in our study believed that wild parrots were important for nature and that they themselves had a responsibility to protect parrots. Local people do not want parrots to disappear and are strongly opposed to outsiders coming in to their community to catch them. On the contrary, up to 74% per community agreed that it was OK to take parrots from the wild to sell, and up to 52% believed that it was OK to keep them as a pet. Furthermore, up to 30% disagreed that catching parrots could make them locally extinct. We found similarity between attitudes and reported behaviours. In areas with more pet *A. lilacina* and reports of parrot trapping, there was also a stronger belief that this was acceptable, compared to areas with fewer pets and trapping. This suggests that changing these attitudes could have an impact on future behaviour and that implementation of a targeted behaviour change education project could have conservation benefits to *A. lilacina*. We suggest following the practices of the successful PRIDE campaigns (Butler et al. 2013) which inspire people to take pride in the species and habitats that make their communities so unique, whilst introducing viable alternatives to environmentally destructive practices.

Chapter Seven: Final conclusions and recommendations

With this research I aimed to further the understanding of the conservation status, distribution, fine-scale habitat preferences, roosting dynamics, and human interactions with a newly described Amazon parrot *Amazona lilacina* (del Hoyo & Collar 2014). The key findings suggest that the global population is between 741-1,090 birds, and that a decline of 60% in the last 19 years in some parts of the range, justifies an IUCN Red List status of Critically Endangered (Biddle et al. 2020; BirdLife International 2020b). My distribution models show that there is 17,772 km^2 of suitable habitat for the species, which overlaps with a high proportion of Ecuador's human population (INEC 2010). I report that *A. lilacina* uses at least four tree species for nesting, five for roosting, 33 for feeding, and has preferences for certain fine-scale habitat characterises suggesting habitat specialisation, which may put the species at an increased risk of extinction. I estimate the size of one subpopulation, which uses a well-known traditional roost site in the south of the range, at 149 individuals - considerably lower than estimates from twenty years ago (Berg & Angel 2006). I also report a significant change in the seasonal roosting dynamics of this subpopulation and suggest this may be caused by degradation of suitable habitat in the local area. Local trapping and pet keeping is reported as the greatest threat to parrots worldwide (Berkunsky et al. 2017). My community surveys indicate that between 2014 and 2017, up to 50% of community members had owned a pet *A. lilacina*, and up to 26% had trapped one, which is likely to cause additional pressure to the population.

My results provide an overview of the ecological but also the social status of *A. lilacina*. The inclusion of people living within the species native range, in the collection of data, has allowed us to build good working relationships with these communities, particularly those nearby to important *A. lilacina* roost sites. The results presented here will be shared with these communities through the development of new teaching materials and the delivery of a conservation education programme. A recent review of over 100 conservation education programmes worldwide, showed that 98% produced at least one positive outcome from changes in awareness and knowledge, through to direct improvement of ecological indicators (Ardoin et al. 2020). I have shown that the inclusion of local communities in ecological research is extremely valuable, particularly for rare species that are hard to study in the wild (Biddle et al. 2021). I have developed and raised funds for a ‘community guardian’ project that will allow us to continue research and protection activities, such as roost monitoring, species observations and nest guarding, throughout the species range. Many case studies show that community engagement can be highly effective in tackling wildlife crime, through the development of shared goals for management of natural resources (Roe 2015) and I believe this approach is key to the conservation of *A. lilacina.* My results indicate an urgent need for a collaborative approach to conservation; with governments, local NGOs and conservation organisations working together to enforce law, reduce trapping, and to ensure vital remaining fragments of forest are protected, but most importantly, for local communities to be engaged and empowered towards the conservation of this species.

It is suggested that *A. lilacina* could be the most threatened mainland Amazon parrot globally (Pilgrim 2010). There are 48 parrot species in Ecuador, ten of these occur on the coastal plain, of which, three are threatened with extinction; the Great Green Macaw *Ara ambiguus* (CR), the Grey-cheeked Parakeet *Brotogeris pyrrhoptera* (EN), and the Ecuadorian Amazon *A. lilacina* (CR) (Freile & Restall 2018; BirdLife International 2020b). There are four parrot species, similar in size to *A. lilacina* with overlapping ranges, but none of these are listed as threatened with extinction: Blue-headed Parrots *Pionus menstruus* (LC), Bronze-winged Parrots *P. chalcopterus* (LC), Red-masked Parakeets *Psitticara erthrogenys* (NT) and Southern Mealy Amazons *A. farinosa* (NT) (BirdLife International 2018d; Freile & Restall 2018). This raises the question of why is *A. lilacina* so rare, when these other four species are not. Whilst *A. lilacina* is restricted to elevation <700 m, the other four similar species range up to 1,400 m or have additional areas of range outside of the coastal plain (Ridgely & Greenfield 2001a). The other four species also do not seem to have such complex habitat needs, and are more frequently reported in agricultural and human influenced lands. For example Bronze-winged Parrots *P. chalcopterus* and Red-masked Parakeets *P. erthrogenys* will feed in gardens, maize crops and fruiting trees in agricultural land, and Blue-headed Parrots *P. menstruus* can persist well in fragmented forest and secondary woodlands making them 'less effected by deforestation' according to Ridgely & Greenfield (2001). *Amazona lilacina* appears considerably more shy and cautious than the other four species; during fieldwork, the most frequently observed species was the Red-masked Parakeet *P. erthrogenys*, which was also highly notable in towns and cities. On the contrary *A. lilacina* were particularly hard to observe, and are reported as 'inconspicuous during most of the day remaining in leafy canopy and only revealing their presence by occasional dropping of fruit pieces' (Ridgely & Greenfield 2001a). As a consequence of this shy behaviour, caution over using agricultural land, and restriction to areas <700 m, it is likely that *A. lilacina* has been more heavily impacted by the vast clearance and urbanisation of the lowland coastal provinces than the other similar parrot species.

The Southern Mealy Amazon *A. farinosa* is probably the most morphologically similar species to *A. lilacina* on the coastal plain, it is also restricted to areas <700 m, yet it outnumbers *A. lilacina* in most of its western distribution apart from within the Guayas province (it is also found on the east of the Andes) (Ridgely & Greenfield 2001a). A clear difference between the two species, is that Southern Mealy Amazons *A. farinosa* rarely gather in large aggregations - they are more typically seen in groups of no more than 4-8 birds - whilst *A. lilacina* are reliant on communal roosting behaviour (Ridgely & Greenfield 2001a; Berg & Angel 2006). This communal roosting behaviour makes *A. lilacina* much more restricted by the distribution of suitable habitat than other species, thus contributing to its rarity. Not only do they require the presence of traditional roost sites large enough to accommodate in some cases hundreds of birds, they also have a preference for the rare mangrove ecosystem, and safety from predation (Ridgely & Greenfield 2001a; Berg & Angel 2006; Hamilton & Casey 2016). Furthermore, their roost site must be within a reasonable daily traveling distance of dry forests with suitable feeding and nesting tree species (Ridgely & Greenfield 2001a; Berg & Angel 2006). The population of *A. lilacina* is reported to have been significantly reduced as a result of vast habitat clearance in coastal Ecuador and large volumes of illegal parrot trapping and export (CITES 1986). The resulting small population is now likely to be very susceptible to relatively low-level pressures including small-scale land use change, which may result in loss of feeding and nesting trees through accidental fire or conversion to agriculture, and removal of chicks from nests to keep as pets, or trapping groups of birds at roost sites to sell. This emphasises the need to involve local communities in conservation planning.

Despite the key findings of my research, there are still many questions that need to be addressed in order to help define conservation priorities for this species. For example, a full assessment of remaining suitable habitat would help to confirm the presence of any additional areas *A. lilacina* are using. Recently, there has been a large rise in *A. lilacina* observations recorded on eBird due to an increased local awareness of the species. The total number of records has doubled in the last two years and this offers a fantastic resource for distribution investigations. In terms of population trend, replication of the three-year roost surveys to assess seasonal fluctuation and size, combined with long-term surveys would allow us to determine if the declining population trend seen at the El Salado Mangrove roost site is mirrored across the range. Furthermore, I suggest that the species occurs in three fragmented sub-populations, but in order to understand the implications of this, investigations into the genetic variation between parrots at different roosting sites is essential. In particular, this would help to identify whether birds at the extreme North and South of the range are isolated or if there is evidence of movement between roosting grounds. Locating active nests has been particularly difficult. An important next step would be to confirm if the reproductive timeline I observed in the Cerro Blanco Forest is the same across all parts of the range, or if chicks are fledging earlier in the South as reported by some communities. Finally, further investigation into community reports of parrots nesting in mangrove trees in the far South of the range is needed, although I had no observations of this myself.

It is clear that the Critically Endangered *A. lilacina* should be a conservation priority. From my results, it is possible to draw a number of conclusions and suggestions for the allocation of conservation resources, in the broad areas of habitat protection, habitat improvement, and reduction in trapping, which I expand on in the next sections.

Habitat protection

Habitat protection through the creation of legally protected areas is recognised worldwide for its benefits to biodiversity conservation (Leverington et al. 2010). For example, bird abundance and species richness is shown to be higher in legally protected forests than in exploited forests (Cueto & De Casenave 2000). This is probably due to reduced disturbance – indeed in Ecuador it has been shown that deforestation rates are lower inside protected areas compared to outside (Van Der Hoek 2017). Although 20% of Ecuador is currently included in the national protected areas management system (Negru et al. 2020) very little of this falls within the lowland coastal provinces (Cuesta et al. 2017) where *A. lilacina* occurs. It is suggested that in Ecuador, the effectiveness of protected area management could be improved by increasing the involvement of communities and indigenous people in the decision making and benefits sharing (Negru et al. 2020). In 2008, Ecuador initiated the Soci-Bosque programme, a system that provides direct economic incentives to owners of land with native forests to guarantee its protection over the medium-to-long-term. Areas were prioritised due to their (1) deforestation threat; (2) importance for ecosystem services and; (3) poverty levels (de Koning et al. 2011). This resulted in a large proportion of the lowland coastal areas being listed as high priority. I suggest that support of local communities following a similar system to the Socio–Bosque programme, to provide economic incentives to land owners for the protection of stands of trees that are important traditional *A. lilacina* roost sites, feeding, and nesting grounds, would be beneficial to its conservation. There are also a number of privately protected areas within the coastal provinces, for example the Cerro Blanco Forest (Horstman 2017) which is very important habitat for the species. Often the NGOs that manage these areas lack resources, and therefore I recommend support of these NGOs to ensure effective management and protection of important parrot habitats.

Mangroves are also an important ecosystem for *A. lilacina;* the coastal populations are distributed only where the three isolated patches of mangrove forests remain (Hamilton & Casey 2016). Between 1969 and 2006 there was a 30% reduction in the area covered by mangroves in Ecuador, which coincided with the growth of the shrimp farming industry (Rodríguez 2018). Studies have shown that 80% of the loss of mangrove carbon in Ecuador is due to direct displacement of mangroves by shrimp farms (Hamilton & Lovette 2015). Although all mangroves are protected according to Ecuadorian law, 58% of the human population lives within 100 km of the coast, and depends on services provided by marine and coastal ecosystems (Rodríguez 2018). As of 2018, 46.5% of mangroves were classed as protected areas and 42% were protected through sustainable use and custody agreements or mangrove concessions, leaving 11.5% with no protection (Rodríguez 2018). Custody agreements or mangrove concessions enable ancestral communities to be granted the sustainable use of mangroves for subsistence (mainly collecting and selling of fish and crabs). The Socio-Manglares system aims to provide economic incentives to associations that have such agreements, supporting them to implement mangrove management plans, which contain guidelines on the sustainable use of resources, control and surveillance, and monitoring and evaluation. A review of these management plans suggests that there may be lack of resources and/or ability of crab fisher and fisher associations to implement plans, and that technical assistance provided by NGOs and universities, amongst others is very important support (Coello et al. 2008). I suggest that working with crab fisher and fisher associations may help to ensure that the importance of mangroves to *A. lilacina* is recognised and built-in to mangrove management plans. Furthermore, that additional resources and support to these associations outside of protected national parks, for example to assist with surveillance, may help to ensure protection of *A. lilacina's* traditional roosting grounds.

Habitat improvement

A recent study has shown that the most negative predictor of health of Ecuadorian dry forest habitat, is the human pressure index, which measures things such as wood extraction, grazing and trampling (Cueva Ortiz et al. 2019). My research suggests that outside of privately protected areas, there may be a lack of frequency and diversity of parrot feeding and nesting trees. I therefore suggest that increasing the number of parrot feeding and nesting tree species, through community initiatives in communal or Socio-Bosque forests, may be beneficial to *A. lilacina.* By providing support to enable and facilitate community reforestation projects, using the 36 “parrot friendly trees” identified in this research, it could be possible to increase the amount of resource available to parrots, whilst engaging community members in parrot protection within communally owned lands that parrots use. I also recommend provision of support to encourage the use of sustainable agriculture techniques, which in the long term may reduce the area of communally owned forest that is cleared for agriculture and improve the quality of remaining forest. Reduction in the use of chemical inputs can lead to an increase bird populations (Stoate 2002; Stevens & Bradbury 2006) which is also likely to improve overall forest quality. A number of NGOs have expertise in sustainable farming practices in the rural coastal provinces, so supporting them to build capacity in these areas is essential. Another risk to the Ecuadorian dry forest habitat is fire (Horstman 2017). The Guayaquil fire department reported 88 significant fires between 2017 and 2019, of which 60% were caused by humans (I. Solis-Ponce *pers. comms.* Benemérito Cuerpo de Bombero de Guayaquil - Captain Roberto Jurado). In the rural communities, there were reports that such fires can be caused by people burning piles of rubbish in the forest, due to a lack of efficient waste collection services. I therefore suggest that support for correct waste management, litter clear-ups, and fire management training may reduce the risks of fire to the important areas for *A. lilacina*.

My research also indicates that the rate of breeding, for *A. lilacina* attending the El Salado Mangrove roost site may be low, as locating nests within a protected area over many years, was extremely challenging. One possible reason for this is a lack of suitable nesting cavities. Provision of artificial nest sites for birds is a commonly used conservation tool but for parrots, they have shown varying levels of success. For echo Parakeets *Psittacula eques* in Mautritius, they have been highly successful with 41 of 65 nest boxes used in two years, and 71% of these fledging chicks (Tatayah et al. 2007). Also for Scarlet Macaws *Ara macau* the use of nest boxes resulted in an increase in population size from 185-225 to 225-265 in three years (Vaughan et al. 2005). On the contrary, for Yellow-eared Parrots *Ognorhynchus icterotis* at five sites in Colombia, artificial nest boxes were rarely used by parrots (Salaman 2006). Different designs have been used, most commonly PVC tubes or wooden boxes, but in some cases reinforcement of natural cavities has been more successful than the installation of boxes; for example for the Yellow-shouldered Amazon *Amazona barbadensis* in Venezuela, only one artificial box was used compared to all of the 15 repaired natural cavities (Sanz et al. 2003). We therefore recommended continued efforts to locate suitable nesting cavities and active nests within the Cerro Blanco Forest, so that cavity reinforcement can be provided if necessary, and the implementation of artificial nest box trials using designs successfully used by other Amazon parrot species, to supplement nesting opportunity. It is important to note that provision of nest boxes that may be more accessible than natural cavities, may result in increased risk of disturbance. Four out of five *A. barbadensis* clutches in artificial nest boxes were removed by poachers (Sanz et al. 2003) and three Scarlet Macaw *A.*

macau clutches in PVC tube nest boxes were destroyed by monkeys (Vaughan et al. 2003). It is therefore essential to provide increased resources for ongoing protection and monitoring of nest sites, through ranger presence, patrols and possibly remote monitoring to ensure the safety of chicks if artificial boxes are used.

Reduction in trapping

My research highlighted that trapping of *A. lilacina* for local pet keeping is a threat to the species. Up to 52% of people believed it was acceptable to keep pet parrots and up to 74% that it was acceptable to catch parrots to sell. It has been shown that anti-poaching efforts in the form of additional human presence, can benefit bird populations (González 2003; Vaughan et al. 2005; Granadeiro et al. 2006) and that recruitment of young people (who may be facilitators in parrot trapping) from the local community to act as nest monitors, can significantly decrease poaching rates (Briceño-Linares et al. 2011). A literature review revealed that some form of parrot nest protection reduced nest poaching by up to 50 times (Pain et al. 2006). In the last two years, we have initiated an Amazona Lilacina Community Guardian scheme, where local community members are trained and employed to monitor and protect nest and roost sites, amongst other research activities. We recommended the continuation and expansion of this scheme. In the last three years, we have also been working with local Ecuadorian educators to develop resources and deliver environmental education workshops in communities within the species range, as the lack of environmental education in Ecuadorian schools has been reported as a barrier to reducing parrot ownership and capture (Ortiz-von Halle 2018). Evaluation of this project has shown that after attending workshops, participants have a significantly increased knowledge and awareness of *A. lilacina,* how to recognise it, its main threats, and its endangered and endemic status. I recommend the continuation of environmental education and increased partnerships with local teachers to create lesson plans and resources that allow sessions about the importance of parrots, and dry forest and mangrove ecosystems, to be delivered in schools. Conservation education should be aimed at changing attitudes towards parrot ownership / trapping, and combined with increased protection of wild birds through nest and roost guarding. Finally, during community surveys it was suggested that there are economic drivers of parrot trapping. First, that farmers have a need for additional income in March to pay for school uniforms and equipment at the start of the school term in April. Second, that crab fishers are legally prohibited from catching and selling crabs in March and consequently may trap and sell parrots instead. I recommended that research into these possible economic drivers of parrot trapping is needed, with the aim to develop solutions that reduce the need for the supplemental income that it provides to families.

It is clear from the results presented in my thesis that local communities believe wild parrots are important for nature and that they themselves have a responsibility to protect nature. Local people do not want parrots to disappear from their local areas and they must be supported to enable them to protect *A. lilacina* from future extinction.

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Conservation status of the recently described Ecuadorian Amazon parrot *Amazona lilacina*

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Summary

Amazona lilacina is a threatened species endemic to Ecuador, existing across a patchwork of mangroves, lowland coastal forests, agricultural and community owned land. The species was described in 2014 and listed as 'Endangered' on the IUCN Red List, however, full assessment of the population was lacking. Using a combination of field observations, roost surveys and community questionnaires, conducted over the last 20 years, we provide up-to-date information on the species' Extent of Occurrence, estimate its global population size, and evaluate its level of threat. Our results suggest the species occurs across an area of 19,890 km^2 in three distinct geographically isolated subpopulations. Roost surveys across the range estimate the minimum remaining population at 741–1,090 individuals and we present evidence to suggest a 60% decline over the past 19 years in one part of the species' range. We conducted community questionnaires with 427 people from 52 communities. The presence of pet parrots was reported in 37 communities, including 17 communities which reported pet *A. lilacina*. From this we predict that over half of all communities within our study area keep parrots as pets and at least 96 communities keep *A. lilacina*. Our findings justify an IUCN Red Listing of at least 'Endangered' for this species and highlight the need for conservation support. In order to assess population health in more detail, further research is required to assess genetic diversity and roost dynamics, and to identify areas that may be important for feeding and nesting throughout the range. As many of these areas are likely to overlap with community owned land, we suggest that future conservation actions should revolve around, and be led by, these communities.

Resumen

Amazona lilacina es una especie amenazada y endémica a la República del Ecuador. Habita en un mosaico de manglares, bosques costeros, predios agrícolas y tierras comunitarias. La especie fue descrita en el año 2014, e incluida en la Lista Roja de la UICN bajo la categoría 'En Peligro', sin embargo, no se realizó la evaluación completa de la especie. Mediante información derivada de observaciones en terreno, monitoreo de dormideros y encuestas a comunidades, actividades realizadas durante el transcurso de los últimos 20 años, aquí presentamos un informe actualizado de la Extensión de la Presencia de la especie, una estimación del tamaño poblacional global, y una evaluación del nivel de amenaza que enfrenta. Nuestros resultados sugieren que la especie ocurre a lo largo de un área de 19,890 km^2, en tres subpoblaciones que se encuentran geográficamente aisladas. El monitoreo de los dormideros a lo largo de su área de distribución dan una población mínima estimada de 741 - 1,090 individuos, y presentamos evidencia que sugiere que la población en una zona de su rango distribucional ha disminuido en un 60% durante el trascurso de los últimos 19 años. En encuestas realizadas con 427 personas de 52 comunidades la presencia de loros como mascotas fue reportado en 37 comunidades, incluyendo 17 comunidades donde se registró la presencia de ejemplares de *A. lilacina* en cautiverio. A partir de esto, predecimos que existen loros en cautiverio en más de la mitad de las comunidades dentro del área de estudio, y que en por lo menos 96 de éstas hay *A. lilacina.* Nuestros resultados justifican la inclusión de la especie, como mínimo bajo la categoría 'En Peligro' de la UICN, y destacan la necesidad de esfuerzos para su conservación. Con el objetivo de evaluar la salud poblacional en mayor detalle, se requieren más investigaciones para evaluar la diversidad genética y la dinámica de los dormideros**,** y para identificar áreas que pudiesen ser importantes para alimentación o nidificación a lo largo de su rango distribucional. Ya que es altamente probable que muchas de estas áreas coincidan con zonas pobladas, proponemos que acciones para la conservación de la especie a futuro deberán centrarse en, y ser lideradas por, estas comunidades.

Keywords: amazon parrot, Ecuador, Endangered, pet trade

Introduction

A third of all Psittaciformes are classified as threatened and over half of all populations are in decline, yet population parameters to support conservation status assessments are missing for many Neotropical parrots (Berkunsky *et al.* 2017). Parrots are commonly threatened by habitat loss, persecution, and the pet trade, and many species are in need of conservation support (Berkunsky *et al.* 2017). Endemic parrots with small fragmented populations, and those such as Amazon parrots with large body size and long generation times, are disproportionately at greater risk of extinction (Snyder *et al.* 1987, Purvis *et al.* 2000, O'Grady *et al.* 2004, Olah *et al.* 2016). Indeed 58% of species in the genus *Amazona* are currently listed by the IUCN as threatened or 'Extinct in the Wild' (BirdLife International 2017).

Amazona lilacina is endemic to Ecuador and was described as a full species in 2014 (Pilgrim 2010, del Hoyo and Collar 2014). An initial Red List assessment categorised it as 'Endangered' due to its small and fragmented population (BirdLife International 2014) however, detailed status information was lacking, uncertain or outdated. For example, the northernmost limit of the species' extent of occurrence (EoO), was historically recorded as south-west Colombia (e.g. Juniper and Parr 1998, Forshaw 2010), which is now believed to be incorrect. Additionally, its dispersal area and habitat preference were recorded as regions encompassing both mangrove and lowland coastal forest habitats (e.g. Ridgley and Greenfield 2001a, Athanas and Greenfield 2016), yet a recent study confirmed the presence of a large roost in a non-mangrove habitat (Blanco *et al.* 2016).

Evidence suggests population size and trajectory are strongly correlated with extinction risk among vertebrates (O'Grady *et al.* 2004) but, since *A. lilacina* was described as a full species, the population size has not been estimated and little is known about its trend in recent years. The species was reported to have undergone severe population decline prior to the mid-1980s in response to ongoing habitat loss and trapping pressure (e.g. CITES 1986, Ridgley and Greenfield 2001b), and by 1998 the population was estimated at just 400–600 individuals (Juniper and Parr

1998). However, this estimate is now almost 20 years old and its reliability is questioned due both to possible declines and to the recent identification of new roosts (Blanco *et al.* 2016, authors' pers. obs.). Roost surveys have been used to estimate global and local population sizes in many parrots species (e.g. Gnam and Burchsted 1991, Martuscelli 1995, Matuzak and Brightsmith 2007, Dénes *et al.* 2018) and provide a tool for long-term population monitoring (e.g. Wermundsen 1998, Wright *et al.* 2019). *Amazona lilacina's* communal roosting behaviour thus allows us to update the population estimate and conduct long-term monitoring to assess population trajectory.

In response to the 'uplisting' of this species to 'Endangered' in 2014, we re-examined its Red List status through personal field observations and collation of information from local experts, NGOs, and communities, over a seven year period to fulfil four objectives:

1. update the current known Extent of Occurrence and estimate area of daily dispersal;
2. estimate global population size;
3. determine change in roost size as an indicator of overall population trend;
4. quantify prevalence of pet parrots within the species' range.

Methods

Study area and roost sites

Amazona lilacina is reliant on lowland coastal forests (Ridgley and Greenfield 2001b) where it feeds on a variety of fruits and seeds, and nests in cavities formed in the trunks and branches of tree species such as pigío *Cavanillesia platanifolia* and ceibo *Ceiba trichistandra* (Kunz 1996, Berg and Angel 2006). Although we know little about this species' reproductive behaviour, adults appear to explore cavities in October/November and produce one or two chicks that fledge between mid-February and late-March (Kunz 1996, Berg and Angel 2006). As with several other *Amazona* species, with the exception of breeding birds, or at least females during the incubation and early chick stages, it returns to communal roost sites every evening e.g. *A. brasiliensis* (Cougill and Marsden 2004), *A. auropalliata auropalliata* (Matuzak and Brightsmith 2007), and *A. amazonica* (de Moura *et al.* 2012). For *A. lilacina*, these roost sites mainly occur on mangrove islands where birds join together every night (Berg and Angel 2006). Birds tend to arrive at sunset, flying in loose-knit flocks made of paired birds, single birds, triplets or small groups, often making loud contact calls as they fly. For this reason roost locations are often well known by local communities, who hear the birds as they arrive and depart the following morning. In contrast, during the day, birds are secretive and extremely difficult to locate as they feed silently and high in the canopy in small groups (Ridgley and Greenfield 2001b).

For this study, we identified four roost sites that are occupied throughout the year. We believe they contain a large proportion, if not all, of the remaining global population of this species and they are separated from each other by at least 50 km (Figure 1). Roost 1 is located on a mangrove island in Manabí Province and was brought to our attention by Fundación Jocototo in 2012. Roost 2 is located in Santa Elena Province and is known to us through the work of Guillermo Blanco and José Tella (Blanco *et al.* 2016). This is the only roost we know of that does not occur in mangroves. Instead, the birds roost in stands of the locally known 'algarrobo' tree *Prosopis julifora.* Roost 3 is perhaps the most well-known roost, located north-west of the Gulf of Guayaquil, in the El Salado Mangrove Reserve where mangrove islands have been frequented by *A. lilacina* since at least the early 1990s (Berg and Angel 2006). Roost 4 is situated south-east of the Gulf of Guayaquil on an island within the Manglares Churute Ecological Reserve. It was located in 2016 through our community questionnaires.

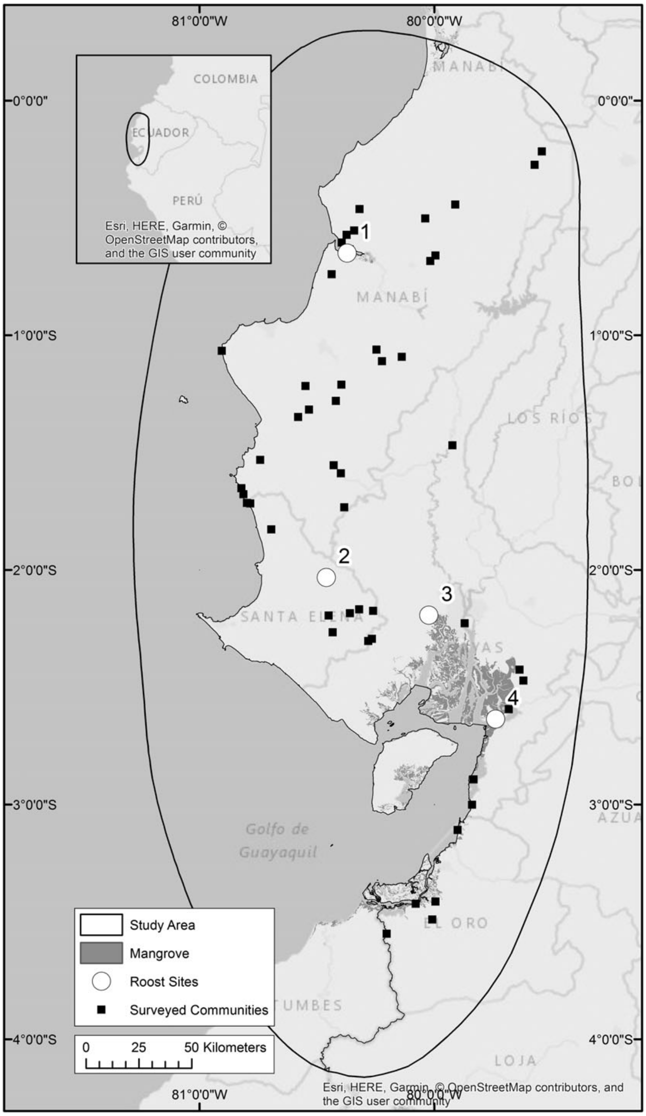

Figure 1. Four *A. lilacina* roosts are believed to contain the majority of the global population, three of these occur on mangrove islands. Mangroves (Hamilton and Casey 2016) and communities taking part in researcher-led questionnaires are indicated.

Field observations

Observational data collected during 10 field trips were used to address Objective 1 (November 2012, January and August 2014, November 2015, August 2016, January and March 2017, February 2018, January and August 2019). Field trips lasted 2–3 weeks during which we investigated potential areas of suitable habitat, verified any recorded sightings of individuals, and monitored known and newly reported roosts. Data collection was informed by: 1) existing information on known distribution and habitat use (Juniper and Parr 1998, Ridgley and Greenfield 2001b, Berg and

Angel 2006, Forshaw 2010, Athanas and Greenfield 2016); 2) information on habitat distribution from Google Earth and available ecosystem maps (Ministerio del Ambiente 2012); 3) direct communication with local NGOs, ornithologists, local guides and bird tour companies and 4) communication through researcher-led questionnaires with local communities.

All sightings of perched *A. lilacina* made by us, Fundación Jambeli staff, and Juan Freile within the last 10 years were georeferenced (sightings of birds in flight were omitted). eBird presence data were lacking, however complete checklists that failed to report *A. lilacina* were used to gain an idea of absence areas: a total of 34,974 complete checklists for mainland Ecuador were downloaded in February 2019.

Roost surveys

To meet Objective 2, we conducted repeat surveys at all roosts. Although these were not located through systematic survey, they represent the combined current knowledge regarding this species according to the authors, local experts and communities. Initially, we conducted practice censuses at each roost to identify the best vantage points. Surveys were then conducted twice per day and where possible, for a minimum of four days to control for intrinsic variability (minimum of 2, maximum of 20 consecutive surveys). To maximise our chances of counting all individuals leaving or arriving at each roost, morning surveys began before sunrise and lasted for two hours, whilst evening surveys began an hour before sunset and finished when it was too dark for birds to be identified. To reduce observer bias, all surveys were carried out by a combination of the same three researchers (RB, ISP, PC), with one person counting and identifying birds using binoculars, the other keeping record. Roost sites are separated by at least 50 km and it has been suggested for other amazon species that if roosts sites are isolated by > 8 km, daily movement between roosts is unlikely (Cougill and Marsden 2004). Still, to account for possible movement of birds between roosts, which could result in counting the same birds twice, only roost surveys conducted during the same weeks of each year were used to estimate population size. Unfortunately, Roost 1 was disrupted and not occupied by amazons during one year of the study, thus an average of counts before and after this disruption, but prior to the next global count, was used. The sum of these counts is presented as an estimated range in minimum global population size during the given time frame. Counts conducted in March are likely to include both adult and juvenile birds returning to the roost after the breeding season, so are suggested to be the most inclusive estimate.

Surveys from 2014 onwards at Roost 3 were conducted from an observation tower within the town of Puerto Hondo, approximately 300 m in front of the roost, allowing a full view of each parrots' flight path to and from the roost. This tower gives a good view of the roost area and approximately 1.2 km on either side. Morning surveys were conducted, by the same researcher, who attended the vantage point from 05h30 to 07h30. At this roost, birds are only seen flying in one direction (into or out of the roost) and therefore it is unlikely that birds were double counted. A consecutive day counting regime was used - the last four days of each month, which has been previously found to be more precise than counting on random days throughout the month; the regime used in 1999/2000 (Berg and Angel 2006, Cougill and Marsden 2004).

To facilitate Objective 3, we compiled all available surveys conducted at Roost 3 to assess long-term change in the size of this roost over time. Survey data were available from June 1999 to May 2000 (conducted by Berg and Angel, 2006) and for various months between November 2015 and May 2018.

Community questionnaires

To address Objective 4, information on the presence of pet parrots was gathered through researcher-led questionnaires in 52 communities within the study area (Figure 1). A total of 427 people took part, representing between 4 and 23 households per community. 'Open Street Map' (OSM) was used to categorise communities as hamlets, villages, or towns. Communities were selected due to their close proximity to lowland dry tropical forests (Ministerio del Ambiente 2012). Following trial surveys, questionnaires were carried out from January to July 2017. A combination of photographs, questions and sound recordings were used to ascertain if the participant could correctly identify *A. lilacina.* Participants were then asked: *"Are there any pet parrots in your village?"* and *"Which parrot species are kept as pets?"*

Questionnaires were conducted in Spanish and only the researcher (ISP) and participant were present. Due to potential bias in self-reporting behaviour using direct questioning, especially in cases where that behaviour is illegal (Fisher 1993, Nuno and St John 2014), we only asked participants to report the presence or absence of pets in their community as a whole. Participants could decline to contribute and were asked for verbal consent prior to participation once the purpose of the research was explained. Interviews were anonymous and data were coded to ensure that no individuals could be identified.

Data analysis

For Objective 1, observation locations were used to estimate the Extent of Occurrence (EoO) using the IUCN Red List guidelines (IUCN Standards and Petitions Subcommittee 2016). ArcGIS was used to calculate the EoO, defined as "the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of current occurrence of a taxon (IUCN Standards and Petitions Subcommittee 2016). The 'Minimum Bounding Convex Polygon' tool was used within ArcToolbox to estimate area of EoO, with no exclusion areas.

To estimate the area of land that birds are likely to disperse over daily, buffers of 10 km were created around observation points; this is suggested to be the approximate diurnal ranging area of *A. auropalliata* in Costa Rica (Salinas-Melagoza *et al.* 2012). Buffers were dissolved in ArcToolbox. To analyse possible movement between daily dispersal areas, absence points were created using eBird complete checklists that did not record the species. Data were filtered and extracted using the *auk* package in R and following suggestions on best practice from Johnston *et al.* (2019), by restricting checklists to < 5 h duration, > 5 km in length, and with < 11 observers.

For Objective 3, count data from roost surveys conducted using comparable methodology were analyzed to assess any change in the size of Roost 3 from 1999/2000 to 2017/2018. For this analysis, only morning counts were used owing to the conclusions of Berg and Angel (2006) who found that their morning counts were more consistent, larger, and thus more accurate. Additionally, Cougill and Marsden (2004) showed morning counts to be more precise for estimating size of other amazon roosts. A generalised linear mixed model (GLMM) with a Poisson distribution and 'month' as a random effect was fitted to compare counts from the two data sets. All statistical analyses were conducted in R (version 3.6.0; R Core Team, 2019).

For Objective 4, 'Open Street Map' (OSM) was used to identify all communities in the study area, in the categories of hamlet/village/town. Predictor variables were calculated for each community (surveyed and not surveyed) using the 'Euclidean Distance' and 'Values to Points' tools in ArcToolbox. These related to species availability, accessibility and land protection status: distance to nearest sighting/roost, elevation (Jarvis *et al.* 2008), distance to nearest road (defined by OSM), and inclusion status within the National System of Forest and Protected Vegetation 2015 (defined by Ministerio del Ambiente). Additionally, mean Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product, MOD13A3, averaged across the period 2010–2015, was included as a proxy of vegetation cover. Random Forests (Breiman 2001) was used to classify surveyed villages with and without pet parrots, and with/without pet *A. lilacina.* The 'predict' function in this package was then used to predict the likelihood of pet parrots and pet amazons being present in the remaining non-surveyed communities within the study area. Communities with a predicted vote score of 0.6 or over, thus a greater than 60% probability, were classed as likely to have pets.

Results

In total 132 occurrence points were gathered, and analysis of eBird checklists resulted in confirmation of 4,626 points of species absence (Figure 2a). The estimated Extent of Occurrence is 19,890 km² within which 5,313 km² is used by the species during daily dispersal. According to the IUCN's definition of subpopulations (IUCN Standards and Petitions Subcommittee 2016) we suggest that *Amazona lilacina* occurs in at least three distinct subpopulations separated by a minimum of 40 km (Figure 2b).

Minimum and maximum counts from each roost survey (Table 1) reflect fluctuations in the number of birds attending each roost during the survey period. Although it is always possible that more roosts exist within the study area, we believe we have identified all remaining large roosts (> 30 individuals) and thus we estimate the remaining global population at 741–1,090, which includes mature and immature birds. We suggest that counts conducted in March (1,090) at the end of the breeding season, represent the population including young birds, and that counts from January (804) represent the population without breeding birds or at least females with eggs or chicks in the early developmental stages. We saw a slight decrease in global population size between March (1,090) and August (1,046) which may represent juvenile mortality.

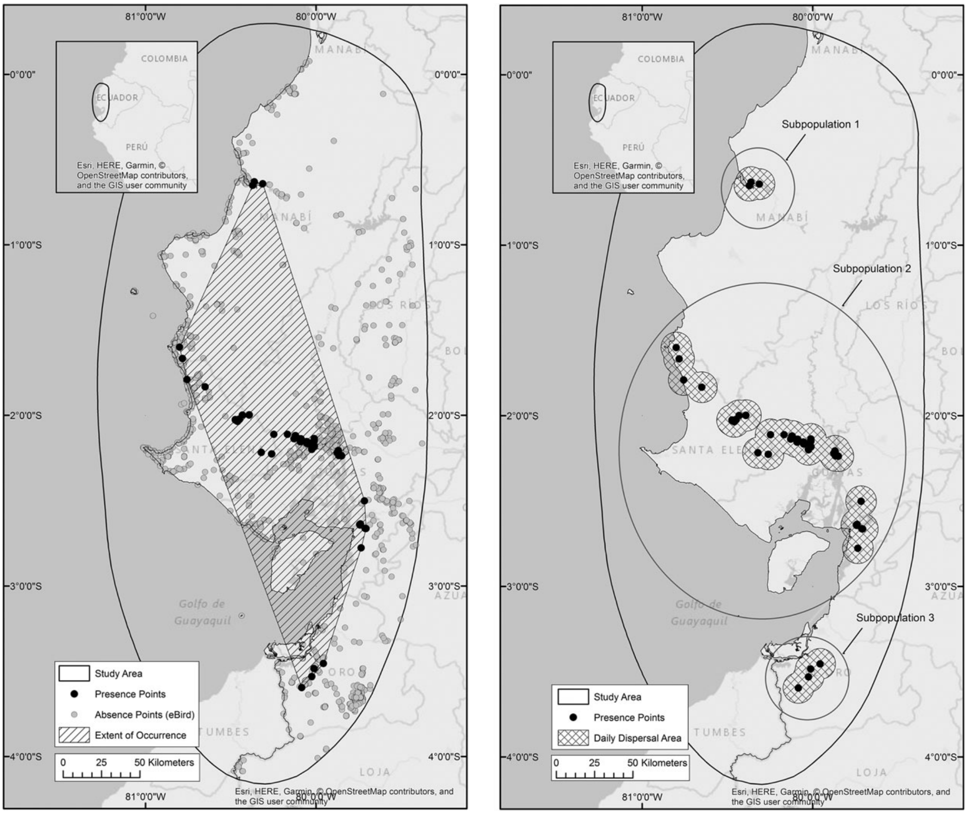

Figure 2a. Presence (*n* = 132) and absence (*n* = 4,626) points recorded for *A. lilacina*. Presence points are joined by a convex hull to estimate the species' Extent of Occurrence of 19,890 km². Figure 2b. Occurrence points are surrounded by 10 km buffers to represent a daily dispersal area of 5,313 km², within three subpopulations.

Table 1. Number of *A. lilacina* counted during roost surveys at all known roosts. Roost 1 was not present in March 2017. Local reports suggest this was in response to damage caused to the mangrove island roost site by a large earthquake. Thus, the mean of all counts prior to January 2019 is used: 97 and 103 (Nov 2012), 84 and 86 (Aug 2014). The population is estimated at a minimum of 741–1,090 individuals.

	Roost 1	Roost 2	Roost 3	Roost 4	Population estimate
March (2017/18)	93	300–480	28–110	320–407	741–1,090
Jan (2019)	44–95	181–338	41–72	220–299	486–804
August (2019)	60–116	124–144	32–57	628–729	844–1,046

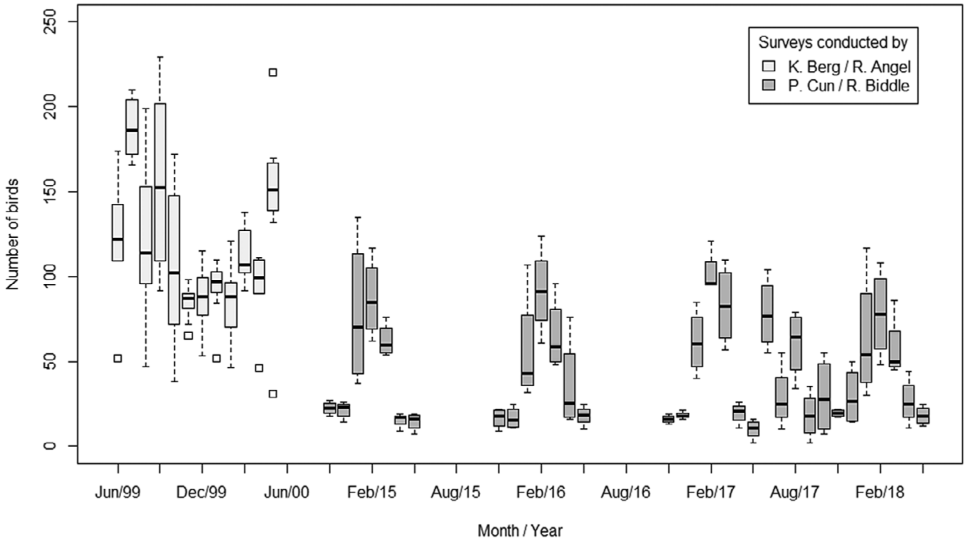

Figure 3. Roost surveys recording the total number of birds departing Roost 3 during morning surveys conducted between June 1999 and May 2018. The average number of birds counted decreased by 60% between 1999/2000 and 2017/2018.

When considering all surveys conducted at Roost 3 from 1999 to 2018, there has been an overall decline in the total number of birds (Figure 3). Our GLMM revealed that average counts declined between the two periods of data collection representing a significant drop in roost size in 2017/2018 compared with 1999/2000 (β = -1.02, SE = 0.24, $P < 0.001$). On average, the difference between monthly counts suggests a 60% reduction in the size of the roost. The maximum roost size in 2000 was 229 birds (Berg and Angel 2006), but just 117 in 2018.

Of 52 communities surveyed, presence of pet parrots was confirmed in 37. A total of 11 parrot species, including *A. lilacina* were recorded: Yellow-crowned Amazon *A. ochrocephala*, Orange-winged Amazon *A. amazonica*, Southern Mealy Amazon *A. farinosa*, Red-lored Amazon *A. autumnalis salvini*, Bronze-winged Parrot *Pionus chalcopterus*, Blue-headed Parrot *P. menstruus*, Red-masked Parakeet *Psittacara erythrogenys*, Grey-cheeked Parakeet *Brotogeris pyrrhoptera*, White-winged Parakeet *B. versicolurus* and Pacific Parrotlet *Forpus coelestis*. Communities with pet parrots could be classified (out of bag error rate 16%) using variables of elevation and distance to the nearest roost, it is predicted that 1,617 of the 3,231 additional non-surveyed communities within the study area have a greater than 60% probability of containing pet parrots.

Of the 37 communities with confirmed presence of pet parrots, 17 held pet *A. lilacina*. These 17 could be classified (out of bag error rate 31%) using predictors of distance to roost, distance to sighting and NDVI. It is predicted using this classification, that 79 of the 3,231 additional non-

Table 2. Of the 52 surveyed communities, 37 reported pet parrots and 17 reported pet *A. lilacina*. Using random forests to predict the occurrence of pet parrots throughout similar communities within the study area, we suggest 1,617 communities have pet parrots and at least 96 have pet *A. lilacina*.

		Pet parrots				Pet *A. lilacina*			
Surveyed communities with confirmed presence		37 / 52				17 / 37			
Out of bag error rate of classification of surveyed communities		15.7%				31.4%			
Classification error table		Confusion matrix:				Confusion matrix:			
			N	Y			N	Y	
		N	10	5	0.33	N	31	4	0.11
		Y	3	33	0.08	Y	12	4	0.75
Mean decrease in accuracy value for each predictor	Community type	2.51				-5.82			
	Distance to sighting	9.45				-1.50			
	Distance to roost	12.95				9.79			
	Distance to road	1.74				-0.88			
	Elevation	16.97				-3.14			
	NDVI	1.98				1.06			
	In protected area or not	4.81				0.17			
Number of communities predicted >60% probability of pets		1,617 / 3,231				79 / 3,231			
Suggested number of communities with pets		1,645				96			

surveyed communities within the study area have a greater than 60% probability of having pet *A. lilacina*. It was felt the value of 60% across the community as a whole, would equate to a much higher probability of at least one pet being owned. From the above, we suggest that within our study area, approximately 1,645 communities have pet parrots, and at least 96 of these have pet *A. lilacina* (Table 2).

Discussion

We estimate the Extent of Occurrence (EoO) for the recently recognised *Amazona lilacina* to be half of that currently listed on the IUCN Red List (BirdLife International 2018), which from available data represents the smallest remaining EoO of any 'Endangered' mainland amazon parrot (Birdlife International 2018). We suggest *A. lilacina* has a population size of between 741 and 1,090 birds and that this population is declining, with Roost 3 showing a reduction in size of 60% over the past 19 years – a similar decline to that seen in other parrot species globally (Berkunsky *et al.* 2017). This rate of decline supports the IUCN listing of 'Endangered' under criterion A, and if reflected over the whole population may qualify the species for listing as 'Critically Endangered'. Further research is needed to assess this, however, when comparing our 2019 counts, to unpublished counts from researchers in 2014, we see a decline of 59% also at Roost 2; an area where strong poaching pressure has been observed (G. Blanco, F. Hiraldo and J. L. Tella pers. comm. 2020). We report that local capture for pets is an ongoing threat and support the notion that Ecuador should be prioritised for parrot conservation (Olah *et al.* 2016).

As commonly seen in parrot roost counts, our results showed variability in roost size within and across months. This may be explained in part by imperfect detection whereby birds arrive at or depart roosts undetected due to low light levels or weather conditions (Dénes *et al.* 2018). Although every attempt was made to account for this, due to the opportunistic nature of some of our roost surveys, a structured counting regime as suggested by Cougill and Marsden (2004) was not always followed. Additionally, it is possible that some birds may have gathered temporarily in smaller, undetected roosts and thus be missed from main roost surveys. Despite this, we believe the results presented here offer a valuable first estimate of population size and trajectory for this 'Endangered' species.

Amazona lilacina's northern border was previously recorded as south-west Colombia or the Esmeraldas province of Ecuador (CITES 1986, Juniper and Parr 1998, Ridgley and Greenfield 2001, Forshaw 2010, Athanas and Greenfield 2016), but we suggest, in agreement with local experts, that these more northern birds are in fact *A. autumnalis salvini* (R. Orrantia, Fundación Jambeli pers. comm. 2013, M. Schaefer, Fundación Jocotoco pers. comm. 2014, R. Ridgley, Rainforest Trust pers. comm. 2015). Within our newly presented EoO we no longer believe the species is restricted to mangrove roosting areas, owing to the discovery of a new roost located > 50 km from any mangroves. However, we do suggest that the species is still highly geographically restricted, with an estimated daily dispersal area of just 5,313 km^2 split between three distinct geographically isolated subpopulations. Although movement between these three areas is unlikely due to their separation distance of approximately 40 km, further research into the daily movement and genetic structure of these subpopulations is needed to confirm this.

Historically, threats to this species have been severe: CITES reported thousands of *A. lilacina* being trapped and exported from the country in the early 1980s (CITES 1986), and Ecuador reported the highest rate of deforestation in South America for the period 2000–2005, with the main cause being clearing of the lowland coastal forests for agricultural crops (Mosandl *et al.* 2008). As early as 1986, the plight of this species was highlighted (CITES 1986) and we believe the population is still at risk and in decline. A likely contributor to this is that the range overlaps with a large proportion (46%) of Ecuador's human population (INEC 2010). In addition to the direct threat of local capture for pets, anthropogenic effects such as fire, hunting, land trafficking and the development of squatter settlements are reported as the greatest threats to the lowland coastal forests this species relies on (Horstman 2017).

We predict that over half of all communities within the study area have pet parrots, despite it being illegal since the mid-1980s to hunt or trade species included in the CITES Appendices. A few of these could be long-lived individuals, however we expect that to be a minority. Ecuador's confiscation reports also suggest a large number of parrots in captivity with 91% of all birds confiscated between 2003 and 2014 being Psittacidae and 7% of these *A. autumnalis* (Ortiz von-Halle 2018). Law enforcement in the form of pet confiscation does not appear to be a strong deterrent, and only once has a case of bird crime resulted in a jail sentence in Ecuador (Ortiz-von Halle 2018). We predict that nearly 100 communities hold *A. lilacina* but expect this is an underestimation due to difficulties in identifying parrots to species level. We did gather evidence of poaching of *A. lilacina* chicks and adults during fieldwork, either to generate core income, or incidentally, to fulfil a specific economic need such as buying uniforms at the start of the school term. Additionally, reports of farmers using nets or poison to protect their crops against parrots and historical reports of family relatives shooting macaws and amazons for food were made. Although legal subsistence hunting does not appear a threat to the species, recent concern has been raised regarding its sustainability in the light of changes in human population size, hunting methods, and habitat fragmentation (Suarez and Zapata-Rios 2019).

The lowland coastal provinces where *A. lilacina* occurs have been identified as having an acute lack of protected areas (Cuesta *et al.* 2017). Additionally, the lowland forests, mangroves and algarrobo trees, are all habitats essential for local community income and sustenance, through hunting, fishing for crabs, cutting of firewood or making charcoal. Outside of these habitats, our observations occasionally recorded the species in crop fields, gardens, and even villages where fruit trees have been planted, and in the last couple of years eBird users are more frequently recording the species within the large city of Guayaquil. This species is clearly existing across a highly anthropogenically-influenced landscape, and although there are examples worldwide of parrots adapting to such environments (e.g. Lill 2009, Martens and Woog 2017) the effects this may have on their natural behaviours or ecological functions could be significant (Luna *et al.* 2018).

Our study highlights an urgent need for a collaborative approach to conservation to reduce *A. lilacina's* vulnerability to extinction; with governments, local NGOs and conservation organisations working together to enforce law and to ensure vital remaining fragments of forest are protected, but most importantly, for local communities to be engaged and empowered towards the conservation of this species.

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ORIGINAL PAPER

The value of local community knowledge in species distribution modelling for a threatened Neotropical parrot

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Abstract

Species distribution models are widely used in conservation planning, but obtaining the necessary occurrence data can be challenging, particularly for rare species. In these cases, citizen science may provide insight into species distributions. To understand the distribution of the newly described and Critically Endangered *Amazona lilacina,* we collated species observations and reliable eBird records from 2010–2020. We combined these with environmental predictors and either randomly generated background points or absence points generated from eBird checklists, to build distribution models using MaxEnt. We also conducted interviews with people local to the species' range to gather community-sourced occurrence data. We grouped these data according to perceived expertise of the observer, based on the ability to identify *A. lilacina* and its distinguishing features, knowledge of its ecology, overall awareness of parrot biodiversity, and the observation type. We evaluated all models using AUC and Tjur R^2. Field data models built using background points performed better than those using eBird absence points (AUC = 0.80 ± 0.02, Tjur $R^2 = 0.46 \pm 0.01$ compared to AUC = 0.78 ± 0.03, Tjur $R^2 = 0.43 \pm 0.21$). The best performing community data model used presence records from people who were able recognise a photograph of *A. lilacina* and correctly describe its distinguishing physical or behavioural characteristics (AUC = 0.84 ± 0.05, Tjur $R^2 = 0.51 \pm 0.01$). There was up to 92% overlap between the field data and community data models, which when combined, predicted 17,772 km^2 of suitable habitat. Use of community knowledge offers a cost-efficient method to obtain data for species distribution modelling; we offer recommendations on how to assess its performance and present a final map of potential distribution for *A. lilacina.*

Keywords Local ecological knowledge · MaxEnt · Psittaciformes · Citizen science · Ecuador

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Introduction

Understanding species distributions is essential for conservation planning (Wilson et al. 2005) but for species that are rare, sparsely distributed, or inconspicuous, this information is often lacking. In such cases, species distribution models (SDMs) and their outputs, can be particularly useful, as long as they are based on ecological theory and built using accurate data (Guisan and Thuiller 2005). SDMs allow the probability of occurrence to be predicted in un-surveyed areas, which can inform future field investigations and have many important conservation applications (e.g. Pearce and Lindenmayer 1998; Araújo et al. 2004). For all SDMs, species presence data are needed. Traditionally this comes from direct species observations or museum records, but more recently scientists have looked to integrate different sources of data, such as citizen science, to make better inferences of the true distribution of species (Amano et al. 2016; Coxen et al. 2017; Fletcher et al. 2019; Steen et al. 2019; Isaac et al. 2020).

The quality of outputs gained from SDMs is affected by factors such as data type, sampling bias and imperfect detection (Lahoz-Monfort et al. 2014; Guillera-Arroita et al. 2015). MaxEnt is one of the most commonly used methods for deriving SDMs and has been shown to produce useful models even when dealing with small sample sizes (Wisz et al. 2008; Elia et al. 2015). Whilst other methods require absence data to be collected, MaxEnt uses presence data combined with a background sample drawn randomly from the study area (Phillips et al. 2006, Phillips and Dudík 2008; Elith et al. 2011). Both presence-absence and presence-background data methods have limitations; namely that presence data often do not represent an unbiased sample of locations at which the species is present, and that absence data can lead to the inclusion of false absences (Guillera-Arroita et al. 2015). These limitations must be considered against the proposed use of model outputs; for instance, presence-background data may be sufficient when outputs are to be used to direct further field investigations, but insufficient if outputs are to directly inform land management for conservation (Lahoz-Monfort et al. 2014). The predictive ability of models may also be reduced if imperfect detection is not accounted for, and may result in outputs being more likely to predict areas in which the species is easier to observe, rather than where it is more likely to occur. It is therefore essential that the effects of imperfect detection are minimised by ensuring a sufficiently large sampling effort at surveyed locations (Lahoz-Monfort et al. 2014).

For species where field observations are lacking, citizen science data is a valuable and widely used resource (Brook and McLachlan 2008) which can help determine species presence, absence or abundance (Melovski et al. 2018; Díaz-Ruiz et al. 2019; Ghoshal et al. 2019; Skroblin et al. 2021). Some methods allow large volumes of data to be collected more cost effectively than traditional field survey methods, for example postal surveys (FitzGibbon & Jones 2006), telephone interviews (Mallory et al. 2003) and social media (Pace et al. 2019). Often this information is used to supplement 'expert' data by guiding further field surveys (Hart & Upoki 1997; O'Brien et al 1998; Chaiyes et al. 2017) but in some cases it is shown to be just as accurate as the equivalent 'expert' data, providing that some form of filter for reliability is incorporated (Polfus et al. 2014). Recently, a number of studies have even shown that georeferenced occurrence data collected through citizen science platforms and online biodiversity databases such as eBird, can be used to build accurate SDMs (Bradsworth et al. 2017; Coxen et al. 2017; Fournier et al. 2017; Saunders et al. 2020). However, it is important to note that all opportunistically collected citizen science data present additional challenges such as spatial biases and variation in observer skill

(Isaac and Pocock 2015; Johnston et al. 2020) and online recording schemes such as eBird create barriers by requiring observations to be collected and submitted in a particular way.

Within all types of citizen science data, there is variation in accuracy. For example studies have shown that 'freelisting' (Bernard 2006), a quick survey method where participants are asked to list the species they see in their local area, can result in people reporting species that do not occur and omitting ones that do (Can and Togan 2009; Díaz-Ruiz et al. 2019). However, the cost efficiency of citizen science may compensate for reduced accuracy depending the data collected and extent of errors (Gardiner et al. 2012). If citizen science data are to be used to infer information about distribution, and as input data for the creation of SDMs, some method of boosting data accuracy or accounting for level of expertise is essential (Kosmala et al. 2016; Johnston et al. 2019). Previous studies have used prior selection of participants i.e. only interviewing key informants selected by community leaders due to their perceived expertise (Mallory et al. 2003; Lopes et al. 2018). Others have developed some kind of scoring system, to determine data accuracy (Frey et al. 2013) by only regarding contributions from participants who are able to recognise photographs of the study species and provide accurate location information (Ghoshal et al. 2019), or by using photographs of non-native species to assess participants identification skills (O'Brien et al. 1998).

To further our understanding of the distribution of a newly described and Critically Endangered parrot species *Amazona lilacina* (Biddle et al. 2020; BirdLife International. 2020), we:

1. Built distribution models using all known locality records of *A. lilacina* from our own observations, those from expert ornithologists, and reliable eBird records (2010–2020);
2. Collected data on local peoples' experiences and observations of wild *A. lilacina* through structured face-to-face interviews;
3. Grouped community interview data based on different quality filters and used these data to build distribution models;
4. Determined the best performing distribution models built from species records and community reports, and compared their outputs in order to direct future field investigation.

Methods

Study area

Amazona lilacina, a species recently split from the *A. autumnalis* group, is found in the coastal region of Ecuador where its small population is sparsely distributed around dry forests and mangrove ecosystems (Biddle et al 2020). These habitats are described as amongst the most imperilled ecosystems on earth (Dodson and Gentry 1991). During the day-time *A. lilacina* is highly inconspicuous, feeding silently in the forest canopy in small groups which presents difficulty in using traditional field survey methods to collect presence data (Ridgely and Greenfield 2001a). However, in the evenings birds will form conspicuous groups and fly to communal roost sites (Berg and Angel 2006) which means that communities living anywhere on this flight path, are often aware of the species presence.

The rural coastal communities are considered to be in the most deprived areas of Ecuador, with almost one quarter of all people living in multidimensional poverty (Mideros 2012). The deprivation gap regarding food and water, education, communication, and housing, is greater here than in any other part of the country (Mideros 2012). Within our sampled communities (Fig. 1a), people mainly make a living as farmers, fishers or crab fishers, and 60% have either none, or only primary level schooling. Many communities in this region are highly inaccessible, especially in the rainy season and 57% of people we surveyed had lived in their village their entire lives. The flow of information into and out of these communities is reported to be infrequent, with only 40% of households having access to one form of telecommunication (radio, television, phone, computer) (Mideros 2012).

Field observations and eBird records

Observational data were collected during ten field trips led by RB, lasting two to three weeks each (November 2012, January and August 2014, November 2015, August 2016, January and March 2017, February 2018, January and August 2019). Data collection was informed by: (1) existing information on known distribution and habitat use (Juniper and Parr 1998; Ridgely and Greenfield 2001a, b; Berg and Angel 2006; Forshaw and Knight 2010; Athanas and Greenfield 2016); (2) information on habitat distribution from Google Earth and the Ministerio del Ambiente ecosystem map; (3) direct communication with local NGOs, ornithologists, local guides and bird tour companies. All sightings of perched *A. lilacina* made by RB, ISP, MP, Fundación Pro-Bosque staff, Fundación Jambeli staff, and Juan Freile between 2010 and 2020 were georeferenced (sightings of birds in flight were omitted).

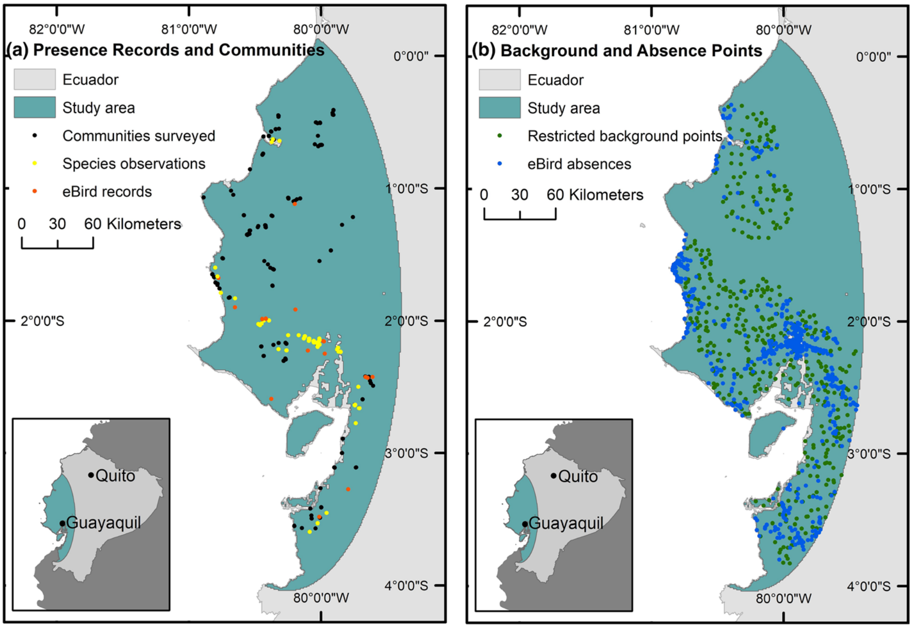

Fig. 1 **a** Locations of all households taking part in interviews, all records of *Amazona lilacina* collated between 2010 – 2020 and, **b** eBird absence points, representing all complete checklists that did not report *A. lilacina*, and random background points matching the number of eBird absence points available, within a 30 km buffer of all *A. lilacina* presence records

All eBird data for Ecuador, including observations and sampling data were downloaded in December 2020. To ensure that no records were missed due to changing taxonomic nomenclature, data were filtered to include all birds recorded as *A. autumnalis* (which included *A. a. lilacina* and *A. a. salvini)* between 01/01/2010 and 31/12/2020. Records that were not deemed as *A. lilacina* based on either photographic evidence or location (i.e. within the Esmeraldas province) were removed, as were records that were already represented by our own observations (within 1 km). To avoid misrepresentation of location, all records that were reported as "general area" which implies the record does not correspond to that exact location were removed, as were records with survey effort > 5 h and > 5 km in length (Johnston et al 2019). Finally, locations of parrots within urban locations in the big city of Guayaquil (visualised on Google Earth) were removed to avoid escaped pets or captive birds being included in models.

Distribution models from field observations and eBird records

The MaxEnt function of the package 'dismo' (Hijmans et al. 2020) in R (version 3.6.0, R Core Team 2019) was used to create species distribution models from field observations and eBird records, referred to from now on as the field models. These were first built using eBird absence points generated by filtering for all complete checklists within our study area that did not report the presence of *A. autumnalis* (*A. a. salvini* or *A. a. lilacina*) (Fig. 1b). Absence points were also limited to checklists that were < 5 km in length, < 5 h in duration and with fewer than ten observers (Johnston et al. 2019), and to a buffer of 30 km from all field observations and eBird records. Our second and third field models were built using random background points generated in ArcGIS (Version 10.8.1) from within the same buffer: the second model had 4597 and the third had the same number as eBird absences available (458). Spatial autocorrelation was controlled for by limiting points to one per 1 km using the R package 'spThin' (Aiello-Lammens et al. 2015). A set of interpolated bioclimatic predictor variables available from WorldClim (https://www.worldclim.com/bioclim) representing different measures of temperature and rainfall, plus additional predictors thought to have some biological significance for the species were used: Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product over the period 2010–2015 as a proxy of vegetation cover; distance to mangrove (Hamilton and Casey 2016) and distance to the nearest river (Military Geographic Institute, IGM). Predictors were checked for pairwise correlation across random points within the study area, using pair plots (Zuur et al. 2010); where correlation coefficients between pairs of predictors were ≥ 0.70, the less biologically meaningful predictor was removed. The final variables were; distance to the mangrove, distance to a river, annual mean NDVI and NDVI seasonality, mean diurnal temperature range, annual mean temperature and temperature seasonality, precipitation of wettest month, precipitation of coldest quarter and precipitation of driest month. To allow comparison between the field and community models, we averaged predictor values across 9 km^2 at all points used in all models to reflect respondents' reference to their 'local area', which could encompass areas of community owned land > 1 km away from their house. To ensure this did not affect model outputs or accuracy we trialled models built using predictor values at the exact location, compared to those averaged over 9 km^2, and found no difference.

Models were evaluated with AUC and Tjur R^2 (Tjur 2009) over five-fold cross validation; the mean evaluation metrics and their standard deviation are presented. AUC measures how well model predictions discriminate between presence and absence (Wisz et al.

2008). Tjur R^2 represents the difference between the mean model value at the presence locations and the mean value at the absence / background locations. All the data were included in the final models. Finally, we present variable importance scores, with permutation values > 10%, with a high value indicating that the model depends heavily on that variable (Phillips et al. 2006) and response plots for the most accurate field model.

Community questionnaires and response filtering

Researcher–led questionnaires were carried out to identify areas that were reported by local people to be occupied by *A. lilacina*. Communities were chosen to be included in this study due to their close proximity to dry lowland forests (within approximately 10 km), identified using the Ministerio del Ambiente ecosystem map. Furthermore, all communities surveyed were inside or within 70 km of the species Extent of Occurrence (Biddle et al 2020). A pilot study was conducted after which interviews were carried out in January-July 2017. Questionnaires were conducted in Spanish by a local Ecuadorian researcher (ISP), with only the interviewer and respondent present (Tourangeau and Yan 2007). We aimed to survey a minimum of three households per community representing a cross section of demographic groups, but often this depended on the availability of participants and the size of the community. In all cases, prior verbal consent was obtained, and although less than fifteen people did not complete interviews, interviewees could decline from contributing once the purpose of the research was explained (Online Resource 1).

The location of each questionnaire, normally by the participant's house, was recorded and participants were asked to respond with reference to their immediate local area which included their house, garden, and local community land. Demographic information regarding age, gender, level of schooling, and how long they had lived in the village, was collected, but interviews were anonymous, and data were coded to ensure that no individuals could be identified. Interviewees were not made aware of the species in concern before starting the interview, during which they were asked to name and describe which parrot species (if any) they see in their local area, then confirm from a selection of ten parrot photographs (the order of which was rotated at random between surveys) (Table 1). If a

Table 1 Photographs of ten parrot species were presented to questionnaire participants

Parrot species	Chance of observation	Species status
1. Southern Mealy Amazon *Amazona farinosa*	Unlikely	Rare
2. Blue-headed Parrot *Pionus menstruus*	Possible	Common
3. Bronze-winged Parrot *Pionus chalcopterus*	Possible	Uncommon
4. Red-masked Parakeet *Psittacara erythrogenys*	Likely	Common
5. Great Green Macaw *Ara ambiguus*	Unlikely	Very rare
6. Pacific Parrotlet *Forpus coelestis*	Likely	Common
7. Grey-cheeked Parakeet *Brotogeris pyrrhoptera*	Possible	Uncommon
8. Lilacine or Ecuadorian Amazon *Amazona lilacina*	Possible	Rare
9. Brown-headed Parrot *Poicephalus cryptoxanthus*	Not present	Not present
10. Yellow-crowned Amazon *Amazona ochrocephala*	Not present	Not present

The chance of observation is rated as likely (if the species range covers the entire study area), possible (if the species range covers more than half of the study area) or unlikely (if the species range covers less than half of the study area) (Freile and Restall 2018) participant confirmed they currently (within the last year) see *A. lilacina* at their location, they were then asked a number of questions designed to help assess the accuracy of this information. Each interview (Online Resource 2) took approximately 20 min to complete.

To examine the influence of accuracy of community data, we filtered responses according to the ability to recognise the species, knowledge of its distinguishing features, overall awareness of parrot biodiversity, and observation type (i.e., if the bird was seen flying, nesting, perched or feeding). We created six groups of responses to represent realistic scenarios that may be used to select which observations to include in distribution investigations (Table 2). We created a further 11 groups which represented all possible combinations of groups three-six, for example group seven represented a group of participants who had answered correctly for all of groups three, four, five and six (Online Resource 3).

Distribution models from community data

We created distribution models based on groups of community data with varying levels of accuracy as listed in Table 2; the community models. Each participant's response was associated with a location representing a 1 km^2 pixel on our distribution maps. These presence locations were combined with environmental variables and background points following the same methods as for the field model. All background points were restricted to buffers of 30 km from community survey presence points. We averaged predictor values across the 9 km^2, as for the field model, to reflect respondents' reference to their 'local area', which could encompass additional areas of community owned land. In order to evaluate the accuracy of the community data models, we use the same methods as for the field models; AUC and Tjur R^2 (Tjur 2009) over five-fold cross validation. We present these, alongside permutation values where their contribution to the model is $> 10\%$ for all models, and the habitat suitability output and response plots for the best performing model.

Model comparison

Once we had identified the best performing field observation model and community data model, we compared the overlap between their habitat suitability outputs. These outputs are interpreted as maps of potential distribution with values indicating the level of habitat suitability for each pixel, on a scale of zero to one. There are several methods used to compare model outputs (Galante et al. 2018). We chose Moran's I which represents the difference between suitability values at each cell, and the relative rank coefficient which estimates the probability that the relative suitability ranking for a patch of habitat cells is the same for the two models (Warren and Seifert 2011). We calculated these using the niche overlap function in ENMTools (Warren et al. 2010). Both methods produce metrics which range from zero (no overlap) to one (complete overlap).

To predict areas of potential distribution, it was necessary to classify areas as either 'suitable' or 'unsuitable' depending on their model value. Many thresholding rules are justified for presence-only occurrence data (Peterson et al. 2011). We chose the 10% omission rate threshold (Galante et al. 2018) where the model value which includes 90% of the values predicted at the presence locations used to create that model, is applied as a threshold to the habitat suitability output to distinguish between presence and absence. We calculated and applied this independently to the two best performing models. We present a final combined map of distribution that represents areas predicted as suitable or not by either of the final models. We extracted the values for the top three predictor variables from the

Table 2 Six methods of community data filtering were developed to group community responses

Community data group	Question	Answer suggesting accurate presence report
1	Can you name the different parrot species you see in your local area?	Gives any name used locally for *A. lilacina*
2	Can you confirm from the following photographs, which parrot species occur in your local area?	Recognises and identifies the *A. lilacina* photograph
3	Describe how you distinguish *A. lilacina* from other parrot species around here	Confirms *A. lilacina* is present from photograph or naming, and also provides one or more physical (description of the overall body shape i.e. square tail, medium size, red forehead, black beak) or behavioural (migration behaviour and associated distinct vocalisation) characteristic specific to *A. lilacina*
4	What habitat do you think *A. lilacina* prefers?	Confirms *A. lilacina* is present from photograph or naming, and also describes preferred habitat as forest or mangrove ecosystem
5	Can you confirm from the following photographs, which parrot species occur in your local area?	Confirms *A. lilacina* is present from photograph or naming, and also recognises the Red-masked Parakeet *Psittacara erythrogenys* and the Pacific Parrotlet *Forpus coelestis* (which are both common and likely to be observed)
6	What were the *A. lilacina* doing in your local area? (flying / nesting / feeding / perched / something else)	Confirms *A. lilacina* is present from photograph or naming, and also confirms it was either perched, nesting or feeding (i.e. not just flying over)

All possible combinations of response groups three-six were included, resulting in 17 groups of community data from which to build species distribution models best performing models, in areas where both models predicted presence, compared to areas where only the field model or only the community model did, and plotted these using the R package ‘ggplot2′ (Wickham 2016).

Predictors of community data performance

Once the best performing community data model been determined, a generalized linear mixed model (GLMM) was conducted in R (version 3.6.0, R Core Team, 2019) using the package ‘lme4′ (Bates et al. 2020). The binomial response of whether or not a participant was included in the response group used to build that model was analysed to determine any effects of participants’ social demographics: gender, level of schooling, age and number of years in the village. Only communities where at least one wild *A. lilacina* observation had been reported were included, and the community location was included as a random effect. We checked for correlation between the age and number of years spent in the village using Pearson’s product-moment correlation, and between gender and level of schooling (some or none) using a Chi-squared test of independence, and only included non-correlated variables in our GLMM.

Results

Field observations and distribution model

Our field observations generated a total of 132 occurrence points. A further 14 locations from eBird were included, to create a final dataset of 146 *A. lilacina* presence locations. These were reduced to 59 (47 field observations and 12 eBird records) during the spatial rarefication process, combined with either: 458 eBird absence points (model 1); 4597 randomly generated background points (model 2) or; 458 randomly generated background points (model 3) and entered into model building with the ten non-correlated predictor variables. The resulting mean of five-fold cross validation AUCs were 0.78 ± 0.03, 0.80 ± 0.02, 0.79 ± 0.02 and the resulting mean of five-fold cross validation Tjur R^2s were 0.43 ± 0.21, 0.46 ± 0.01 and 0.41 ± 0.01 for models 1 to 3, respectively. Therefore, field model 2 was considered to be the best performing model (Table 3). The habitat suitability output from model 2 shows that the suitable habitat follows the Chongón Colonche mountain range, from Guayaquil north-west towards the coast, with additional suitable areas in the far south of the country bordering Peru, and the north of the study area in mid-Manabí (Fig. 2a). Environmental variables that showed a permutation importance of $> 10\%$ were annual mean NDVI, distance to the mangrove, and temperature seasonality and response plots (Fig. 2b) suggest that suitability of habitat is associated with close distance to mangrove and a relatively high annual mean NDVI.

Community questionnaires and reliability scoring

A total of 404 people from 72 communities took part in questionnaires, including 183 women and 221 men, with an average of 5.6 households per community (min 2, max 23). There was a variety of schooling levels, from none (31), primary (214), secondary (128), to university (31) and in how long participants had lived in their community (1–84 years) but the majority (88%) had lived there for ten or more years. Of the

Table 3 Results of the three field models showing permutation values of environmental variables where > 10%

Model	Sample size (after thinning)	Permutation importance of variables > 10%	Mean of five-fold cross validation AUC (± sd)	Mean of five-fold cross validation Tjur R^2 (± sd)
Field model 1 with eBird absences (458)	146 (59)	Mean annual NDVI (34%) Precipitation of coldest quarter (19%) Precipitation of wettest month (16%) Mean diurnal temperature range (14%)	0.78 (± 0.03)	0.43 (± 0.21)
Field model 2 with background points (4597)	**146 (59)**	**Mean annual NDVI (32%)** **Distance to mangrove (24%)** **Temperature seasonality (17%)**	**0.80 (± 0.02)**	**0.46 (± 0.01)**
Field model 3 with restricted background points (458)	146 (59)	Mean annual NDVI (25%) Distance to mangrove (24%) Temperature seasonality (21%) Precipitation of coldest quarter (20%)	0.79 (± 0.02)	0.41 (± 0.01)

Based on AUC and Tjur R^2 values, model 2 (in bold) is the best performing field data model

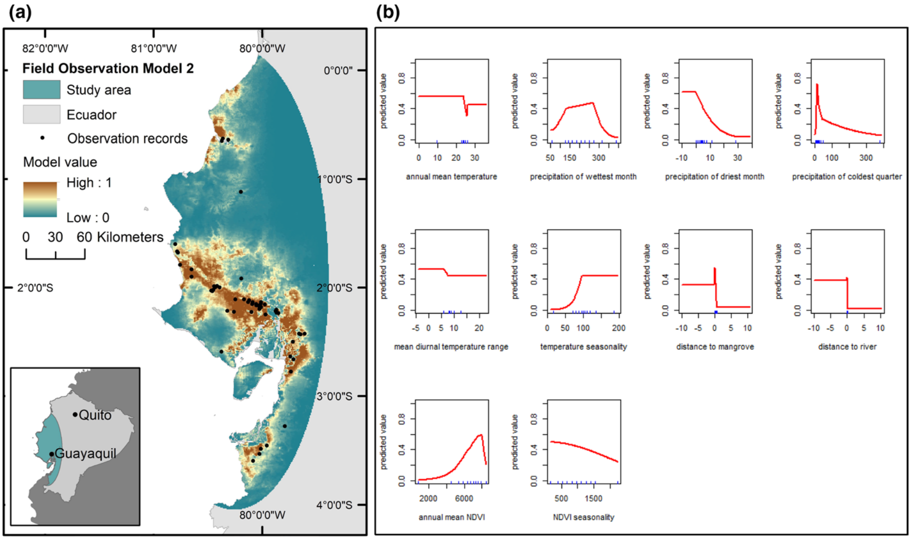

Fig. 2 **a** The habitat suitability output from the best performing field model which is built using 59 species records and 4597 background points **b** The variable response plots for this model

404 participants, 393 reported seeing parrots in general. Although it was posed in our questionnaires that participants should answer with reference to birds seen in the wild, when asked “where did you see this bird?” 15 respondents replied “as a pet” - these 15 responses were removed from the community models.

Distribution models from community data

After filtering community data based on the six groups in Table 1, and creating combination groups where participants answered positively for multiple categories, each group had a sample size of ≥27 (27–155). After spatial thinning all datasets contained ≥18 (18–67) georeferenced occurrence points. Each group of points was combined with 3,931 background points and the same ten non-correlated predictor variables as those included in the field models. Models were built based on groups one to six of data, and then all 11 possible combinations of groups three to six. None of the combination models improved the performance of the model (Online Resource 3). The mean of five-fold cross validation AUC for the six main models was $> 0.74 \pm 0.03$ and Tjur $R^2 > 0.39 \pm 0.02$. Based on these values, model 3 is the best performing community model (Table 4). The habitat suitability map of community model 3 shows a similar area of suitable habitat to the field data model, but with additional increased suitability predicted along the coastline (Fig. 3a). Environmental variables with a permutation importance of > 10% were distance to mangrove and temperature seasonality, and response plots for this model suggest that suitability of habitat is associated with areas closer to mangroves (Fig. 3b).

Table 4 Georeferenced occurrence points for each group of community data were thinned to control for spatial correlation

Model	Sample size (after thinning)	Permutation importance of variables > 10%	Mean of five-fold cross validation AUC (± sd)	Mean of five-fold cross validation Tjur R^2 (± sd)
Community model 1	27 (18)	Distance to mangrove (52%) Distance to river (14%) NDVI seasonality (20%)	0.83 (± 0.11)	0.48 (± 0.06)
Community model 2	155 (67)	Distance to mangrove (35%) Precipitation of wettest month (17%)	0.77 (± 0.10)	0.42 (± 0.03)
Community model 3	**115 (53)**	**Distance to mangrove (54%)** **Temperature seasonality (12%)**	**0.84 (± 0.05)**	**0.51 (± 0.01)**
Community model 4	134 (64)	Distance to mangrove (43%) Precipitation of wettest month (22%)	0.76 (± 0.06)	0.41 (± 0.01)
Community model 5	67 (40)	Distance to mangrove (53%)	0.76 (± 0.07)	0.40 (± 0.03)
Community model 6	40 (26)	Distance to mangrove (37%) Precipitation of wettest month (19%)	0.74 (± 0.03)	0.39 (± 0.02)

The resulting models all showed high accuracy (AUC > 0.74 ± 0.03). Based on the AUC and Tjur R^2, model 3 (in bold) is the best performing community data model

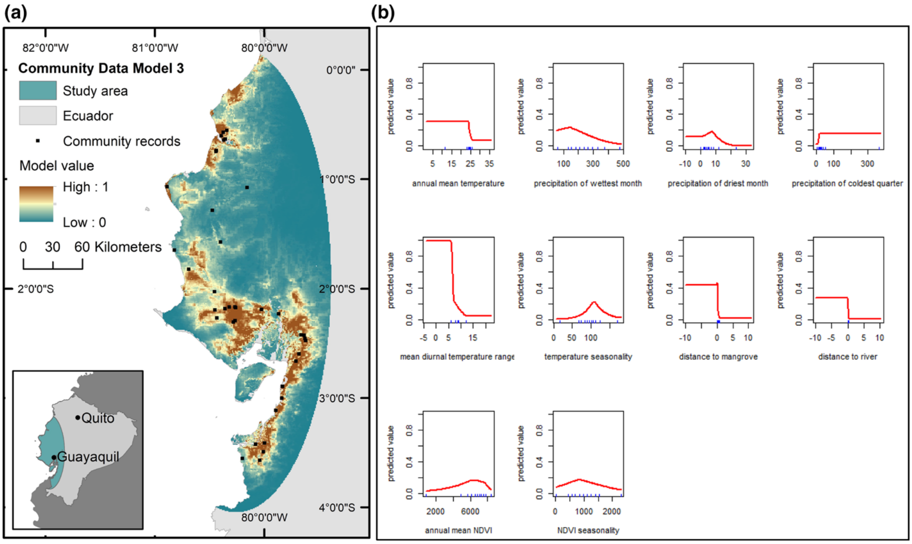

Fig. 3 **a** The habitat suitability output from the best performing community data model, built using 53 reports where participants were able to recognise a photograph of the species and provide one or more physical or behavioural characteristics specific to *A. lilacina.* **b** The variable response plots for this model

Model comparison

After calculating and applying thresholds to the best performing field and community models, the field model predicts 13,969 km^2 of suitable habitat and the community model predicts 13,067 km^2 (Table 5). When we combine these threshold habitat suitability outputs, they overlap in 9314 km^2 of predicted suitable habitat, the community data model predicts a further 3753 km^2 that the field data does not, and the field data model predicts a further 4655 km^2 that the community model does not (Fig. 4). The top three predictor variables from both of these models were; distance to mangrove, temperature seasonality and mean annual NDVI. When plotting the values from predicted presence areas by both models, just the field model or just the community model, areas that are predicted by only the community model have a slightly lower mean annual NDVI and are closer to mangroves than areas only predicted by the field model (Fig. 5). There is a high level of overlap between the field data and community data habitat suitability outputs (before applying a threshold). The relative rank coefficient, which estimates the probability that the relative suitability ranking for a patch of habitat cells is the same for the two models, is 0.82, and the Moran's I, which represents the difference between suitability values at each cell, is 0.92 (Table 5).

Predictors of community data performance

Of the 52 communities where at least one observation of wild *A. lilacina* was made, and thus species presence was likely, 35% (105/304) of participants were included in community data group with the best model performance. These 105 participants (70 men and 35 women) were able to either name or recognise a photo of the species, and describe one

Table 5 The area of predicted presence by the best performing field and community data models after calculating and applying a 10% omission threshold, and the level of overlap between the two habitat suitability outputs before applying a threshold

Most accurate models	10% omission threshold value	Predicted presence area (km^2)	Level of overlap: relative rank	Level of overlap: Moran’s I
Field data model 2	0.30	13,969	0.82	0.92
Community data model 3	0.46	13,067		

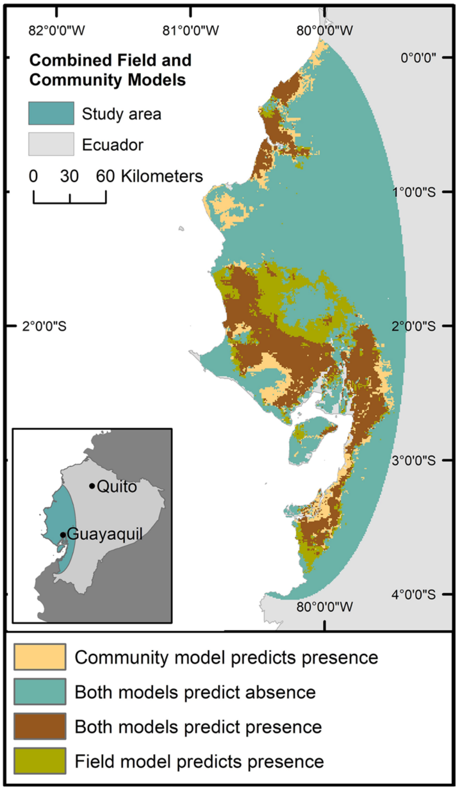

Fig. 4 After calculating and applying thresholds independently to the two best performing models, their predicted suitable habitat overlaps in 9314 km^2, but the community data model predicts a further 3753 km^2 that is suitable, and the field data model predicts a further 4655 km^2 that is suitable for *A. lilacina*

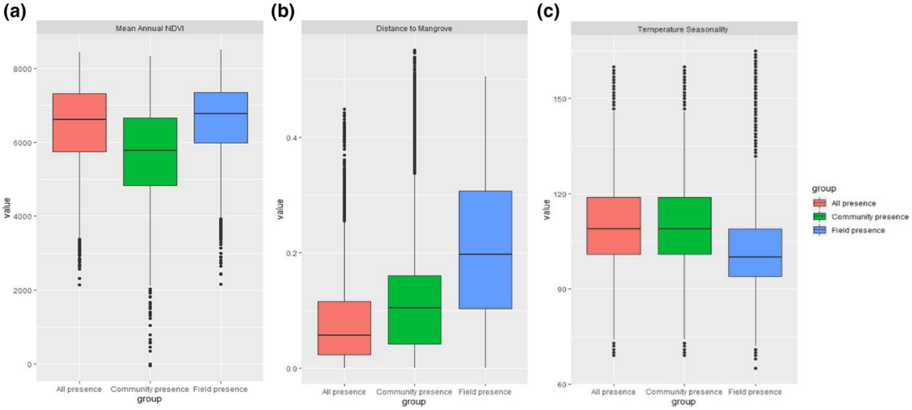

Fig. 5 Box plots showing predictor values in areas predicted as suitable (after applying a threshold) by both the best performing community and field data models, only the field data model, and only the community data model. The predictors with a permutation importance of >10% in the final models were included; mean annual NDVI **a** distance to mangrove **b** and temperature seasonality **c**

Table 6 The gender, level of schooling, and mean number of years lived in the village, for all participants that lived in communities where parrots were likely to be present, and for those who were included in the best performing community data group (group 3)

Response group	Gender	n	Level of schooling		Number of years in the village (mean)
			Some	None	
All responses	Total	304	279	25	34
	Women	138	134	4	30
	Men	166	145	21	37
Community data group 3	Total	105	100	5	36
	Women	35	35	0	32
	Men	70	65	5	38

of its distinguishing physical or behavioural characteristics (Table 6). There was a high correlation coefficient of 0.70 ($p<0.001$) between the number of years lived in the village and the age of a participant. Additionally, gender and level of schooling were significantly correlated ($X^2=8.24$, $df=1$, $p=0.004$). Therefore, we only included the number of years a participant had lived in the village, and the participant's gender in our GLMM. This revealed that of participants living in areas where *A. lilacina* was likely to be present, men were more likely to be included in the better performing community data group than women (Coefficient value: 0.62 ± 0.31, $p=0.04$), which is likely due to their spending more time outdoors in traditionally male working roles. The number of years a participant had lived in the community (Coefficient value: 0.012 ± 0.007, $p=0.14$) had no significant effect.

Discussion

We found that both field data and citizen science data in the form of community surveys were able to produce accurate species distribution models and their outputs had an overlap of 92%. When using field data, we found that models built using background points performed better than those built using absence points generated by eBird checklists, possibly due to the low frequency of eBird records in our study area. When using community data, we found the best performing models were those built using reports from observers who could name or recognise a photograph of *A. lilacina* and correctly describe at least one distinguishing physical or behavioural characteristic.

Recent studies have shown that web-based citizen science projects and online biodiversity databases can be used to build reliable species distribution models (e.g. Saunders et al. 2020; Langham et al. 2015; Fournier et al. 2017). This study presents evidence that in areas where there are substantial barriers to web-based citizen science projects, for example in socio-economically deprived areas (e.g. Hobbs and White 2012), community surveys can overcome these barriers and produce accurate species distribution models. This is of particular use for newly described and rare species. Gender disassociation in local ecological knowledge is not uncommon (Kai et al. 2014; Aswani et al. 2018); we found that men were more likely to provide accurate answers than women and suggest that this is due to a gender difference in traditional working roles (Voeks

2007; Ayantunde et al. 2008) which allows men to spend more time outdoors. Erosion of local ecological knowledge is a global trend (Aswani et al. 2018) and we support the continuation of community wide engagement projects to minimise this risk, with a focus on support for women to enable them to engage with conservation.

After applying thresholds to our best performing field and community data models, they overlapped in their predictions of suitable habitat by 92% (in 9314 km^2). The level of overlap we see between our community and field data models is greater than seen in similar comparison of eBird community data and field-based satellite tracking data of Band-tailed Pigeons *Patagioenas fasciata* (Coxen et al. 2017). Our community data model predicts a further 3753 km^2 of suitable habitat that our field data model does not. These areas were closer to mangroves than areas predicted only by the field data model. This may be due to a factor of species detectability; *A. lilacina* are more detectable (highly vocal) when flying over to mangrove communal roost sites, so perhaps more likely to be seen by local communities in this habitat compared to when they are foraging inconspicuously in the dry forest (Ridgely and Greenfield 2001a). It is also possible that these areas represent locations in which local people have memories of the species occurring in the past, in which they no longer occur and thus were not recorded during field surveys. Our field data model predicts a further 4655 km^2 of suitable habitat that our community data models do not, and in areas with a slightly higher mean annual NDVI than areas predicted only by the community model.

Similarly to Frey et al. (2013), we found variation in the accuracy of community data models built using different methods to filter interview responses. Our best performing model used a filter whereby participants needed to recognise a photograph of the species and provide a reliable description of how they distinguish it from other parrot species in their area. This suggests that, particularly in areas where many similar taxa may occur, the key to assessing the accuracy of information may be simply to ensure that participants are referring to the correct species. This draws parallels with checks that are in place for citizen science online databases such as eBird where records are flagged for systematic review and confirmed by a regional expert prior to their acceptance (Sullivan et al. 2014). It also supports the work of Frey et al. (2013) who conclude that, for easily-identifiable species at least, distribution modelling is possible using anecdotal reports. Our second best community data model (1) greatly underestimated the predicted area of suitable habitat. This group was based on the 'freelisting' method, where participants needed to name the parrot species in their area without any prior information or prompting. Previous studies using the freelisting method have yielded questionable results (e.g. Can and Togan 2009; Díaz-Ruiz et al. 2019) and we believe in our case, it was due to a very small sample size of participants who had the required natural history expertise to name this rare parrot species without any prompting or information.

We found that using identification of other parrot species, to measure overall biodiversity knowledge and therefore accuracy of answers, did not produce the most accurate results. This may be due to *A. lilacina's* unique daily migration behaviour, in some cases flying directly over villages and becoming conspicuous to many community members, not just those that are skilled at identifying multiple parrot species. Alternatively, it is possible that the two parrot species whose identification we assessed as a measure of reliability are incorrectly believed to be common and widespread throughout our study area (Ridgely and Greenfield 2001b; Freile and Restall 2018). Identification of other closely related species was not a good measure of data quality either in surveys investigating the distribution of a native pheasant species – results showed frequent misidentification of an 'imposter' pheasant photograph, but reliable information about the native pheasant was still generated (O'Brien et al. 1998).

Our distribution models based on field data and high quality community knowledge represent the first of their kind for the newly described and Critically Endangered *A. lilacina,* and have important conservation implications. With an estimated population size of just ~ 1,000 birds, and a suggested recent 60% population decline in parts of the range (Biddle et al. 2020), our results have identified new areas to survey. It is important to note that our model predictors did not include factors such as poaching that may have a strong impact on occupancy (Robinson et al. 2010). Whilst conducting community surveys for this study, we discovered a new large roost, unknown previously to local and international ornithologists, located near a socio-economically deprived coastal community, on a mangrove island. Even local residents, because of the conflict with pirates, deem this area as unsafe. We therefore recommend that when parts of a species range fall within areas that are rarely visited by outsiders, the combined knowledge of communities local to that species is likely to be much greater than that of external scientists or researchers, and should thus be used to enhance and supplement traditional field survey methods.

Supplementary Information The online version contains supplementary material available at https://doi.org/10.1007/s10531-021-02169-9.

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Author contributions All authors contributed to the study conception. Material preparation and manuscript writing were conducted RB. Data collection was assisted by ISP, data analysis were performed by RB under the supervision of CD. MJ and SM commented on previous versions of the manuscript and provided conceptual guidance and critical revisions.

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Declarations

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

Ethical approval This study meets full compliance with ethical standards regarding research involving human participants. Full ethical approval for the questionnaires conducted was gained from the North of England Zoological Ethical Review Committee. Questionnaire content and an informed consent statement discussed with each participant prior to interviews, can be found in the online material.

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Article

Parrot Ownership and Capture in Coastal Ecuador: Developing a Trapping Pressure Index

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Abstract: We located rural communities with pet parrots and used these locations to predict the probability of illegal parrot ownership across coastal Ecuador, using variables related to demand for pets, parrot availability, and trapping accessibility. In 12 pet keeping communities, we carried out in-depth interviews with 106 people, to quantify ownership, trapping, and interviewees' attitudes towards these behaviours. We combined these data to calculate a trapping pressure index for four key roosting, feeding and nesting sites for the Critically Endangered Lilacine or Ecuadorian Amazon Parrot *Amazona lilacina*. We found that 66% of all communities had pet parrots and 31% had pet Lilacines. Our predictive models showed that pet parrot ownership occurs throughout coastal Ecuador, but ownership of Lilacines by rural communities, is more likely to occur within the natural distribution of the species. The number of people per community who had owned Lilacines in the last three years varied from 0–50%, as did the number of people who had trapped them—from 0–26%. We interviewed 10 people who had captured the species in the last three years who reported motives of either to sell or keep birds as pets. Attitudes towards pet keeping and trapping differed among the 12 communities: 20–52% believed it was acceptable to keep pet parrots, and for 32–74%, it was acceptable to catch parrots to sell. This being said, most people believed that wild parrots were important for nature and that local people had a responsibility to protect them. We conclude that trapping pressure is greatest in the southern part of the Lilacine's range, and urgent conservation measures such as nest and roost protection, and local community engagement are needed.

Keywords: *Amazona lilacina*; poaching; conservation threats; mangrove; dry forest; local knowledge; attitudes; Lilacine Amazon

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1. Introduction

Parrots (Psittaciformes) are among the most endangered and rapidly declining bird groups, with 28% of their species classified as threatened [1]. Globally, over a third of parrot species are caught to fulfil the demand of the international wildlife trade [1–3]. In the Neotropics, over half of the studied parrot populations are in decline [4], and one reason for this is the high demand for the pet trade [5]. Neotropical species are particularly favoured as pets [2,6], and it is suggested that trapping is a stronger threat to their conservation than habitat loss [7]. Amazon parrots and macaws are preferred due to their attractiveness and ability to mimic the human voice [8]; this is illustrated in Costa Rica, where nearly 20% of households have a pet parrot and half of these are *Amazona* species [9]. Consequently, the rate at which Amazon parrots and macaws are trapped is much higher than expected considering their availability in the wild [10].

Trapping risk is highest where parrots are abundant in the wild, where demand is high and where parrots are relatively easy to catch and sell [11]; therefore, trapping pressure may differ across a species' range and also between species. Additional factors found to drive hunting and trapping include overlap with human population [11,12] and proximity to infrastructure or towns [13]. Attitudes and subjective norms are also factors that influence decision making [14], and are therefore likely to affect the level of pet keeping and capture in different areas. In Ecuador, wild bird keeping is illegal [15], and whilst ownership appears to be declining in major cities [16] demand is still high in rural areas, where over half of coastal communities still keep pet parrots [17]. The most frequently reported confiscated bird species in the country are those with wild distributions exclusive to this coastal region [16,18]. Moreover, this region is one of the most densely populated and impoverished [19] parts of Ecuador, the habitats here have been drastically reduced [20] and are greatly underrepresented in the country's national protected areas system [21].

The Critically Endangered Lilacine or Ecuadorian Amazon *Amazona lilacina*, a species recently split from the *A. autumnalis* group, is found exclusively within the coastal region of Ecuador [22]. CITES reported thousands of individuals of this species being trapped and exported in the early 1980s [23] and although frequency of trapping is likely to have reduced significantly in recent years, there are still multiple reports of capture and pet-keeping within rural communities [17]. An average of 392 wild-caught parrots, including 30 *A. autumnalis*, were confiscated annually in Ecuador between 2003 and 2016 [16]. Although some of these may be older birds, and they may be either *A. lilacina* or *A. a. salvini*, this figure suggests that some level of trapping is still occurring to fulfil the demand for pets. The goal of this study was to understand the risk of trapping in rural communities and formulate a strategy for conservation support. Specific objectives were:

1. Locate communities with pet parrots by conducting surveys across coastal Ecuador, and use these locations to predict the distribution of pet parrots, and the likelihood of local parrot trapping, using variables related to parrot availability, opportunity and demand;
2. Within communities that keep pet parrots, interview local people to quantify the level of parrot ownership, trapping and the attitudes towards these behaviours;
3. Develop a trapping pressure index based on model predictions, locally reported incidence and attitudes towards parrot capture and ownership.

2. Materials and Methods

2.1. Surveys to Locate Communities with Pet Parrots

In order to locate rural communities with pet parrots, we conducted surveys between January and July 2017. The study area encompassed the extent of occurrence of the Lilacine Amazon *Amazona lilacina* and communities close (<10 km) to forest patches, where wild parrots may occur were selected. Participants were asked to confirm if they knew of pet parrots in their community, and if possible to identify the species. Prior verbal consent was obtained from each participant and full ethical approval of survey content and methods was gained from The North of England Zoological Ethical Review Committee. We aimed to survey at least four households per community; however, some communities were made up of just a few houses, so this was not always possible. We recorded the geographic coordinates of communities with all pets, pet parrots and pet Lilacines, and calculated how many communities each species was recorded in. We used IUCN Red List range maps provided by BirdLife International [24] in order to determine if species were native to the study area. Range maps are frequently updated so we report the year of update for each range map in the results. ArcGIS (version 10.8.1) [25] was used, clipping the distribution shape files, to calculate the size of each species range within our study area.

It is illegal to keep native bird species as pets in Ecuador [15], yet in our experience, people speak openly about their parrots and are proud to show them off. However, it was important that participants did not feel threatened or that we were collecting information to inform the authorities. Therefore, surveys were conducted by a local Ecuadorian researcher, in Spanish, with only the researcher and interviewee present, and it was made clear that all information given was anonymous, and only to be used for scientific research.

Although we refer to *A. lilacina* as the Lilacine or Ecuadorian Amazon Parrot, neither of these common names have Spanish translations that are used in Ecuador. Most local communities refer to “loro frentirrojo” (Red-lored Parrot), which in English describes the *A. autumnalis* group and includes *A. a. salvini* in northern Ecuador. To avoid confusion, we use *A. lilacina* in our communication with local communities and use photographs to confirm identity, but refer to the Lilacine Amazon in this manuscript.

2.2. Distribution Models to Predict Parrot Ownership

From our surveys we created two groups of geographic coordinates to represent (1) communities with pet parrots, and (2) communities with pet Lilacines. The MaxEnt package in R (version 4.0.3) [26,27] was used to build distribution models based on these coordinates combined with random background points within 30 km buffers of community locations, to predict the distribution of pet parrots, and the distribution of pet Lilacines. Variables were extracted to match each corresponding location and were chosen due to their influence over parrot ownership and trapping [11]: opportunity (presence of parrots and their desirability); demand (presence of people and the infrastructure for trade); and accessibility (into the forest).

For each location, we calculated a “species value” to represent parrot trapping opportunity. For the pet model, this was calculated based on the presence of wild parrot species at that location using species range maps [24], combined with the frequency of the species being reported as a pet; 0.1 was allocated for each species present in that area, and an additional 0.1 was added if that species was reported in a single community, 0.2 if in two communities, etc. This value was used just for comparative purposes within the study and we gave equal weighting to wild species presence and popularity in captivity, as we had no evidence that either was more important than the other. For the pet Lilacine model, this value was replaced with the predicted occupancy area from our distribution models created using observations of the wild population [28]. For both models, we also used the estimated human population [29], the Euclidean distance to the nearest town and nearest road calculated in ArcGIS using OpenStreetMap [30] data, and the mean annual Normalised Difference Vegetation Index (NDVI) from the monthly MODIS product over 2010–2015 as a proxy of vegetation cover.

For each group of points, spatial autocorrelation was controlled for by limiting them to one per 1 km using the R package spThin [31]. Predictors were checked for pairwise correlation across random points within the study area, using pair plot for collinearity [32]. Model evaluation was performed with five-fold cross validation and the mean AUC +/− SD are presented to demonstrate the predictive ability. An AUC of 0.7 means there is a 70% chance that the fitted model will be able to correctly distinguish between presence and absence [33]. All data were included in the final models. We present the permutation importance (%) of variables, with a high value indicating that the final model depends heavily on that variable [27].

2.3. Interviews to Quantify Parrot Ownership, Trapping and Attitudes

We selected 12 communities where pet parrots were present to conduct interviews with community members about their experiences and attitudes towards parrot ownership and capture. These communities ranged in size from 50 to 300 people. The reason these sites were chosen was because our focus was on understanding risk to Lilacine Amazons, so the selected communities fell within the species extent of occurrence and were <15 km away from key roosting, nesting and feeding grounds [17]. These 12 communities were grouped into four clusters (Figure 1). We interviewed at least six participants from different households in each community. Participants were outdoor workers (i.e., agriculturalists, fishers and crab fishers) selected for their familiarity with parrots in their local area. The same methods regarding informed consent and data anonymity as described in Section 2.1 were followed. Due to low literacy levels amongst participants, all questions were read out aloud and the answer provided was recorded by the researcher. Age and gender of each participant was recorded. The interview consisted of eight questions and seven attitude statements arranged on a five point symmetric Likert scale (Table 1). The Likert package [34] in R (version 4.0.3) [26] was used to visualise attitude statements. Responses were grouped into positive, neutral or negative and a non-parametric test (Kruskal–Wallis) was used to determine significant differences in responses between the four community clusters.

Table 1. Interviews about parrot ownership and capture asking eight questions and seven attitude statements, which were read out aloud by the researcher in Spanish. Interviews were anonymous and participants could decline to answer any questions.

Interview Questions		
1.	How often do you see *Amazona lilacina*?	daily/weekly/monthly/yearly/never
2.	In your opinion, have *A. lilacina* numbers changed in the last three years?	increased/decreased/stayed the same/not sure
3.	Have you ever owned a pet parrot?	yes/no/prefer not to say
4.	Have you ever owned a pet *A. lilacina*?	yes/no/not sure/prefer not to say
	- If yes, how did you get it?	caught it/bought it/given it/prefer not to say
	- If yes, where did you get it?	
	- If yes, how long ago did you get it?	last year/two years/three years/> three years/prefer not to say
	- If yes, how many *A. lilacina* have you owned in the last three years?	
5.	How many other people in the village have a pet *A. lilacina*?	
6.	Have you ever taken *A. lilacina* from a nest or caught one from the wild?	yes/no/not sure/prefer not to say
	- If yes, how many in the last three years?	
	- If yes, for what purpose did you catch it?	
7.	Have you ever sold *A. lilacina*?	yes/no/not sure/prefer not to say
8.	To your knowledge, do other people in your village take *A. lilacina* from nests/the wild?	yes/no/not sure/prefer not to say

Attitude Statements						
Response categories were: strongly disagree (1), disagree (2), neutral (3), agree (4), strongly agree (5), I don't know (NA).	1	2	3	4	5	NA
I think that it is OK to keep a parrot as a pet.						
Catching parrots from the wild can make them extinct in my local area.						
Wild parrots are important for nature.						
I have a responsibility to protect the environment.						
I am comfortable with outsiders catching parrots in my local area.						
It is OK to catch wild parrots to sell to the pet trade.						
It is OK if parrots disappeared from the wild.						

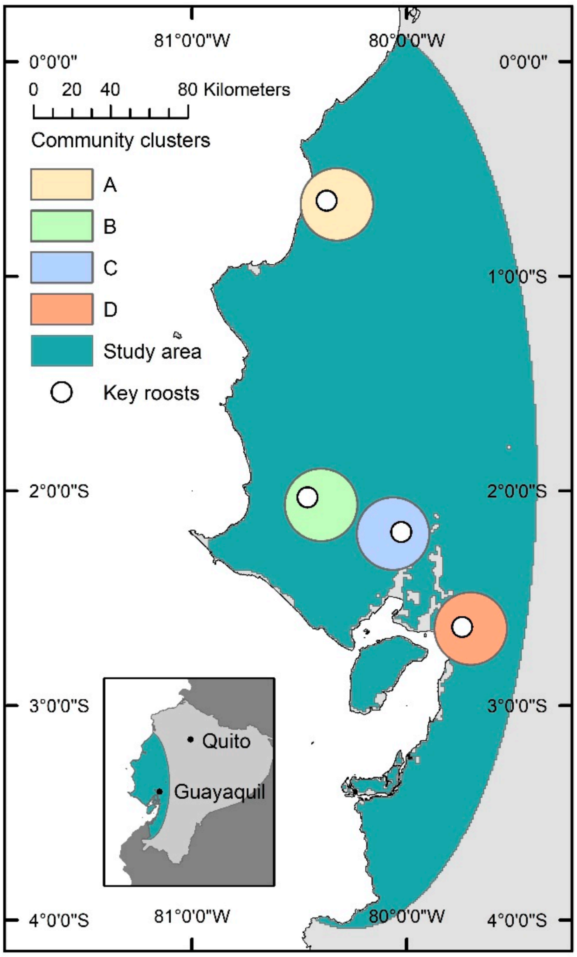

Figure 1. Interviews about parrot ownership and capture were conducted in 12 communities, grouped into four clusters (A, B, C, and D) near key Lilacine Amazon roost sites. Each cluster contains three communities <10 km apart.

2.4. Trapping Pressure Index

To prioritise areas for conservation support, we calculated a trapping pressure value for each of the four community clusters, to represent the level of risk to the wild Lilacine Amazon population from capture and local desire for pet keeping. This risk value was calculated by combining the following six factors: (1) the mean model value for pet parrot keeping (which represents a probability that pets occur at that location); (2) the mean model value for pet Lilacine keeping; (3) the percentage of people who have owned a pet Lilacine in the last three years; (4) the percentage who have trapped Lilacines in the last three years; (5) the percentage of people who think it is OK to keep a parrot as a pet; and (6) the percentage who think it is OK to catch wild parrots to sell.

3. Results

3.1. Locations and Species of Pet Parrots

Surveys were carried out in 65 communities (mean = 6 interviewed people per community; range 3–20). In 43 (66%), pet parrots were confirmed, and in 20 (31%), pet Lilacines were confirmed. Of the 19 wild parrot species, nine were reported in at least one community, with the most frequently reported being Lilacine Amazons (Figure 2) and Grey-cheeked Parakeets *Brotogeris pyrrhoptera*. The mean range size within the study area of parrots found as pets was 27,370 km^2, compared to 8677 km^2 for those not kept as pets (Table 2).

Figure 2. Examples of pet Lilacine Amazons in four rural communities in coastal Ecuador. Parrots were kept in a variety of situations; indoors or outdoors, caged or with clipped wings. In some cases pet parrots that were housed in gardens were not initially considered to be captive by the owner, but for the purposes of this study any parrot living in the locality of people was classed as a pet.

Table 2. The number of communities in which each of the 19 species was reported as a pet and the range size within the study area according to BirdLife International [24].

Parrot Species (Year of Update)	Range within Study Area SA (km^2)	Number of Communities Reporting the Species (Out of 65)
Lilacine or Ecuadorian Amazon *Amazona lilacina* (2018)	38,860	20
Grey-cheeked Parakeet *Brotogeris pyrrhoptera* (2014)	8645	20
Red-masked Parakeet *Psittacara erythrogenys* (2007)	54,327	17
Pacific Parrotlet *Forpus coelestis* (2017)	55,300	8
Red-lored Amazon *Amazona autumnalis* (2017)	5583	6
Blue-headed Parrot *Pionus menstruus* (2013)	27,943	2
Southern Mealy Amazon *Amazona farinosa* (2013)	8612	1
Bronze-winged Parrot *Pionus chalcopterus* (2014)	46,508	1
White-winged Parakeet *Brotogeris versicolurus* (2018)	549	1
Chestnut-fronted Macaw *Ara severus* (2014)	49,329	0
Blue-fronted Parrotlet *Touit dilectissimus* (2014)	13,470	0
White-capped Parrot *Pionus seniloides* (2012)	1482	0
Rose-faced Parrot *Pyrilia pulchra* (2002)	12,828	0
Great Green Macaw *Ara ambiguus* (2014)	3899	0
Red-faced Parrot *Hapalopsittaca pyrrhops* (2000)	49	0
Cordilleran Parakeet *Psittacara frontatus* (2014)	1347	0
Barred Parakeet *Bolborhynchus lineola* (2014)	2183	0
Red-billed Parrot *Pionus sordidus* (2014)	1565	0
El Oro Parakeet *Pyrrhura orcesi* (1999)	615	0
Kept by communities but non-native		
Orange-winged Amazon *Amazona amazonica*	NA	1
Yellow-crowned Amazon *Amazona ochrocephala*	NA	5

3.2. Predicted Distribution of Pet Parrots

The locations of the 43 communities with pet parrots and the 20 communities with pet Lilacines were reduced to 42 and 19, respectively, after limiting each group of locations to one per 1 km. A total of 3803 background points were randomly allocated. The mean AUC of resulting models was 0.69 ± 0.06 (sd) for pet parrots and 0.62 ± 0.20 (sd) for pet Lilacines. The most important variables predicting the presence of pet parrots were distance to nearest road (permutation importance, PI = 40%) and distance to nearest town (PI = 28%); the key factors for the presence of pet Lilacines were the mean annual NDVI (PI = 33%) and species value, representing the native distribution (PI = 27%) (Table 3). Predictions show that pet parrots are likely to be widespread throughout the study area, whereas pet Lilacines seem to be more likely within the species range. Both models show a high probability of occurrence of pets to the west of Guayaquil and out towards the coast (Figure 3).

Table 3. Permutation importance values for variables used to create models predicting the distribution of pet parrots and pet Lilacines in coastal Ecuador.

Variable	**Permutation Importance (%)**	
	Pet Parrot Model	**Pet Lilacine Model**
Mean annual NDVI	11	33
Distance to road	40	1
Human population density	18	23
Distance to town	28	16
Species value	3	27

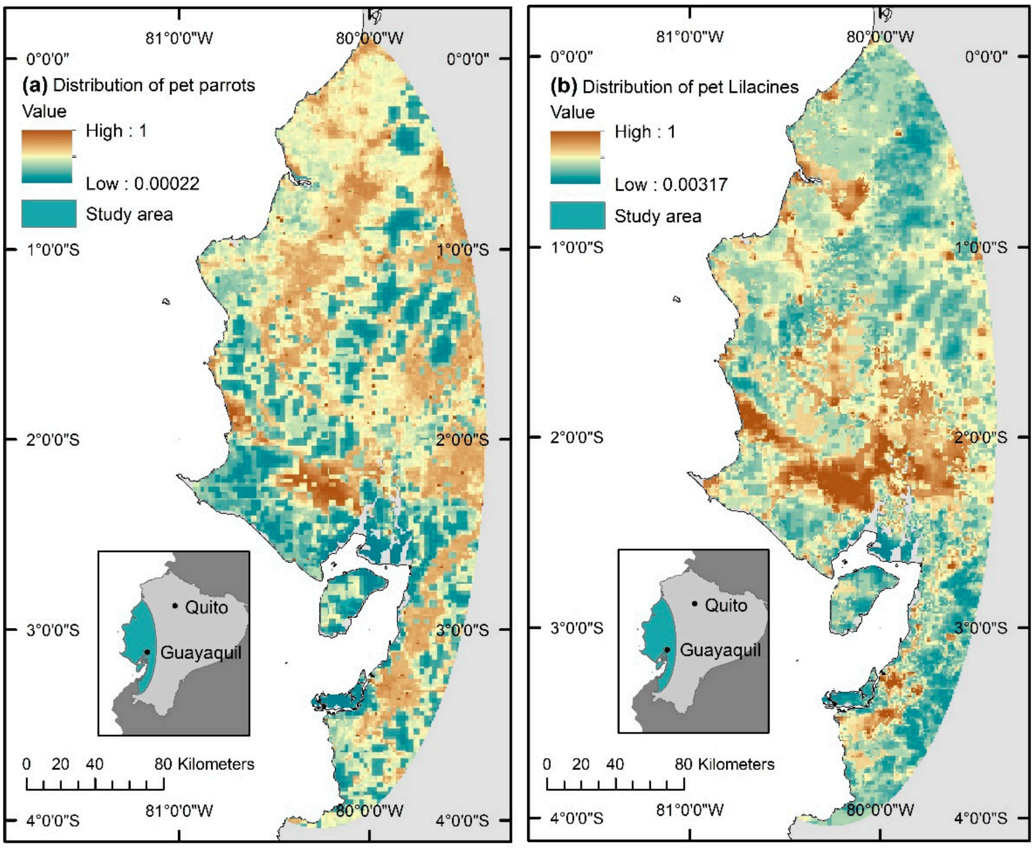

Figure 3. Model predictions showing the distribution of pet parrots (**a**) and pet Lilacines (**b**).

3.3. Incidence of Parrot Ownership and Trapping

Within 12 selected communities where pet parrots occur, 106 (96 men/10 women) participants (min 6, max 13, mean 8.8 per community) took part in interviews. All participants worked outdoors as farmers (57), fishers (25), crab fishers (18), bee keepers (3) or wildlife guides (3). Participants were familiar with *A. lilacina*, the majority seeing them daily (68%), weekly (19%), or monthly (8%), with the remaining 5% just a few times per year. Of all participants, 66% (70) had owned a pet parrot either previously or currently, and 36% (38) a pet Lilacine. The majority (74%, 28) of Lilacine pets had been caught by the owner themselves, with the remainder received as gifts (16%, 6), bought (2%, 6), or found (2%, 6). In the last three years, 15 people have owned a total of 24 Lilacines. In total, 34 people (32%) confirmed that they had previously captured Lilacines, the majority (76%, 26) to keep as a pet themselves, the others to sell (9%, 26) or for undisclosed reasons (15%, 5). Pet ownership and trapping varied between community clusters, with the highest rates of historic and current ownership and trapping of Lilacine Amazons occurring in the crab fishing communities (D) in the southern part of the range (Table 4).

Table 4. The number, age and occupations of people interviewed from each community cluster and the number who reported owning parrots or catching parrots, either previously or in the last three years.

Community Cluster	n	Mean Age (Years)	Occupation: Farmer (F), Fisher (Fi), Crab Fisher (CF), Other (O)				In Life Time:			In the Last Three Years:	
			F	Fi	CF	O	Owned Parrot	Owned Lilacine	Caught Lilacine	Owned Lilacine	Caught Lilacine
A	31	53	8	19	1	3	23	11	9	0	0
B	23	48	23	0	0	0	11	8	8	4	3
C	29	53	26	0	0	3	18	4	4	1	1
D	23	46	0	6	17	0	18	15	13	10	6
Total	106	50	57	25	18	6	70	38	34	15	10

In the last three years, 10 interviewees reported that they had caught Lilacines, with at least 16 birds among them, to either keep the bird as a pet (7), to sell it (1), or for an undisclosed reason (2). All had either no or primary level schooling, and were men 23–72 years old. They reported seeing wild Lilacines daily (9) or weekly (1), and all but one believed the wild population was stable or increasing. In cases where the capture location was given, this always corresponded to the person's occupation, i.e., farmers reported catching parrots in the forest, fishers and crab fishers reported trapping parrots in mangroves (Table 5). Seven of the 10 people who had caught Lilacines in the last three years reported that multiple other people within their community also catch Lilacines, and all 10 knew of multiple pet Lilacines in their community (mean 5.2 Lilacines).

Table 5. The age, gender, schooling, and occupation of all interviewees who reported catching Lilacines in the last three years. We report the trapping location, reason for capture and how many were caught.

Community Cluster	Age (Years)	Gender	Level of Schooling	Occupation	Location of Capture	Reason for Capture	Number of Lilacines Caught in Last Three Years
B	41	Male	Primary	Farmer	Dry forest	Pet	1
	23	Male	Primary	Farmer	Dry forest	Pet	1
	72	Male	Primary	Farmer	Dry forest	Pet	1
C	68	Male	None	Farmer	Undisclosed	Undisclosed	1
D	32	Male	Primary	Crab fisher	Mangrove	Pet	2
	54	Male	Primary	Crab fisher	Mangrove	Pet	1
	40	Male	Primary	Crab fisher	Mangrove	Pet	1
	47	Male	Primary	Crab fisher	Undisclosed	Undisclosed	Unknown
	51	Male	Primary	Fisher	Mangrove	Pet	1
	67	Male	None	Fisher	Mangrove	Sell	7

3.4. Attitudes towards Parrot Ownership and Trapping

Across all communities, responses to attitude statements show a strong feeling that wild parrots are important for nature and participants indicated that local people have a responsibility to protect the environment. This is mirrored by a strong feeling of discomfort with outsiders coming to catch parrots and with parrots disappearing from their area. On the contrary, 46% of all participants believe it is OK to catch wild parrots to sell and 32% that it is acceptable to keep a pet parrot. Furthermore, 17% of people did not believe that catching wild parrots could make them become extinct in the local area (Figure 4). There were no significant differences between communities in the distribution of positive, neutral and negative responses to all attitude statements apart from one: "I think it is OK to keep a parrot as a pet". For this, there was a significant difference between mean responses of the community groups (H = 6.613, $p = 0.022$), with 52% of community cluster D believing this is acceptable, and just 20% of cluster A believing so.

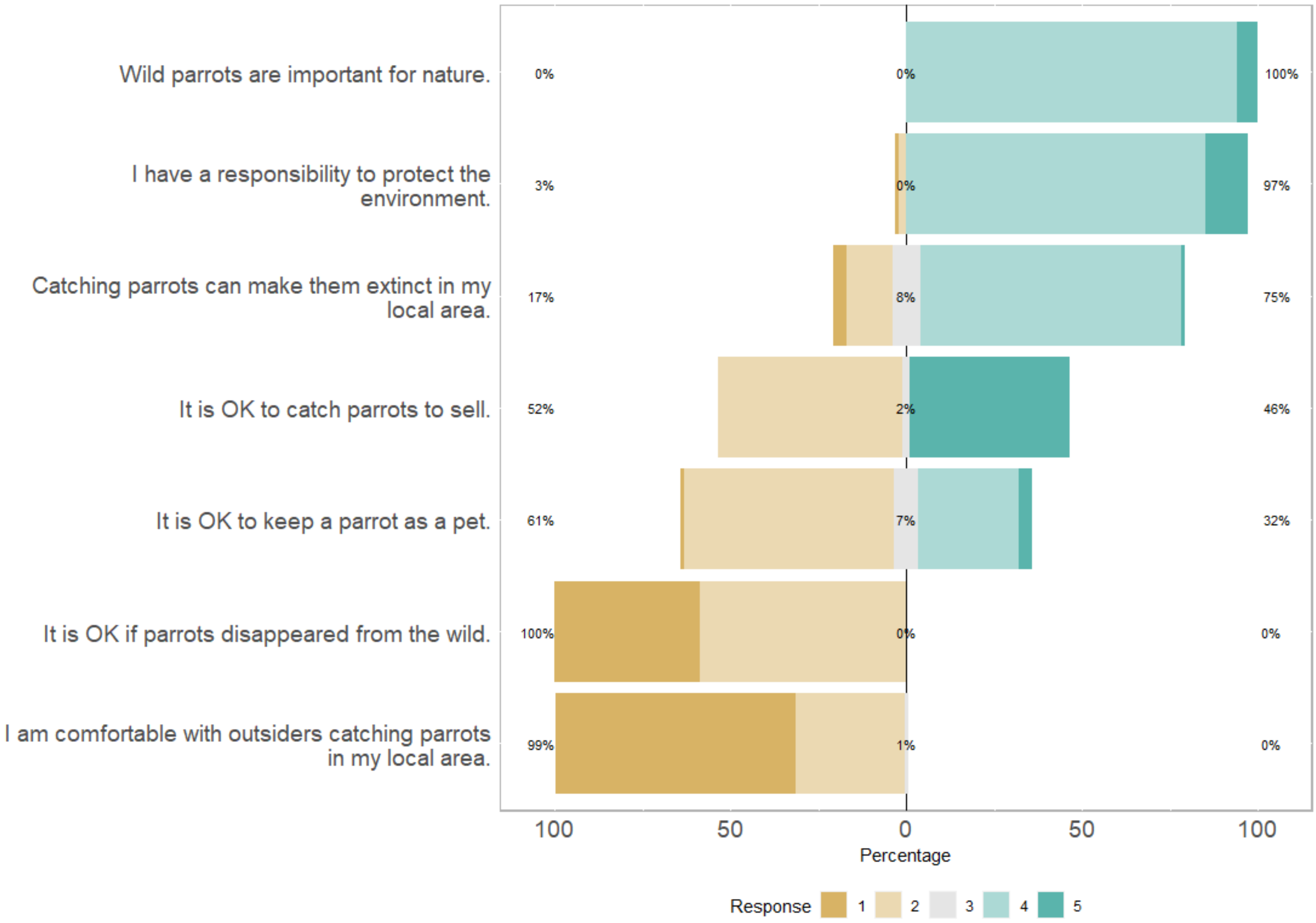

Figure 4. Responses to attitude statements are reported on a five point Likert scale (1 = strongly disagree, 2 = disagree, 3 = neutral, 4 = agree, 5 = strongly agree).

3.5. Trapping Pressure Index

When combining our results into a trapping pressure index, we can see variation between the four community clusters, with higher values suggesting a higher risk to the wild Lilacine Amazon population in that area (Table 6). Wild Lilacines occurring around community cluster D are at greatest risk, due to the high level of reported parrot ownership and capture, and a strong local attitude that this is acceptable. Those occurring around community cluster C are also at high risk, as model predictions here show a high probability of pet Lilacine occurrence (0.78), which suggests a high probability of trapping as pet owners mostly report catching their pet themselves. The wild population occurring around community cluster A appears to be at the lowest risk from trapping, as there were no reports here of current Lilacine ownership or capture; however, this is the smallest remaining subpopulation of the species within its range, which could also explain the low prevalence of pets. When considered against participants' responses to their perceived status of the wild population locally, we see more negative responses from the southern community clusters, with the most frequent response in cluster C being 'decreasing' (76%), in cluster D 'stable' (39%), whilst 'increasing' in cluster B (83%) and A (42%).

Table 6. Trapping pressure index for each community cluster, calculated by adding together factors of predicted or reported level of pet ownership and trapping, and attitudes towards pet keeping, capture and trade. Predicted probabilities were converted into percentages for this calculation.

Trapping Pressure Factor	Community Cluster			
	A	B	C	D
Mean probability of predicted parrot ownership (0–1)	0.47	0.35	0.70	0.36
Mean probability of predicted Lilacine ownership (0–1)	0.51	0.59	0.78	0.19
Percentage of participants who owned pet Lilacines in the last three years	0	17	3	43
Percentage of participants who caught Lilacines in last three years	0	13	3	26
Percentage of participants believing it is OK to keep a pet parrot	20	23	37	52
Percentage of participants believing it is OK to catch wild parrots to sell	39	32	41	74
Overall trapping pressure index (rank)	157 (4)	179 (3)	232 (2)	250 (1)

4. Discussion

This study found that 66% of rural coastal communities in Ecuador have pet parrots and 31% have pet Lilacine Amazons *A. lilacina*. Within these communities, 66% of people had owed a pet parrot during their lifetime, and 14% currently owned Lilacines. This is similar to Costa Rica, where 18% of households owned a pet parrot in 2001 [9]. Our current ownership questions focused on just one species, so we expect the level of current ownership of all parrot species to be much higher and similar to Colombia where 58% of all people had pet parrots [10]. Current ownership and reports of Lilacine trapping in the last three years varied between communities, with 0% to 50% and 0% to 26%, respectively. Ten participants confirmed that they had taken Lilacines from the wild in the last three years to keep birds as pets, or to sell them, so we suggest that, similarly to Yellow-shouldered Amazon *Amazona barbadensis* harvesting in Venezuela, there are at least two categories of trappers—"poacher-keepers" and "poacher-sellers" [35], with only the latter having the contacts and logistics to sell birds. People in occupations with an established and frequent trade link—for example, fishers or crab fishers—may have more opportunity to transport trapped birds to other towns to sell. We also found that capture location corresponded to the occupation of the poacher, which may explain the variation between rural communities as occupation depends heavily on location, i.e., in-land or coastal. Our trapping pressure index identified that the southern distribution of the species is likely to be at greatest risk, which agrees with earlier work showing a vast population decline in this area [17] and provides further evidence that this area should be prioritised for conservation support.

Understanding whether taking parrots from the wild is opportunistic or selective is important because selective capture can lead to the extinction of species through over-harvesting [10]. Our results suggest that trapping is selective given the differences in the popularity of species, with some kept in 20 communities and some in none. The two most frequently reported pet parrot species differed greatly in body mass, which, in general, is linked to longevity in captivity [36], suggesting that variation in popularity is not a side effect of survival rates in captivity. Similarly to previous research, we have noted a preference for Amazon parrots, with all three wild occurring species and two non-native species being kept as pets [8]. However, parrot ownership and capture, at least within rural communities, may also be opportunistic. Most parrot owners had caught their bird locally, within areas they visit during a normal days' work, and our predictions showed that pet Lilacines were more likely within the species' wild distribution. Moreover, parrot species kept as pets had a larger average wild range size than those that are not. This all suggests that ownership and capture are driven in part by parrot availability and accessibility [11], but more research including a true measure of wild parrot abundance, and surveys and interviews in larger towns and cities are needed. According to anecdotal reports in some rural communities, orders are placed by outsiders from cities such as Guayaquil or Quito, which fits the typical multi-level chain involving trappers, middlemen and markets described by Pires [37] and needs investigation.

Our interviews revealed that Lilacine Amazons were trapped both in mangroves, where they roost, and dry forests, where they feed and nest [38,39], suggesting that both adults and chicks are being taken from the wild. Anecdotal reports from communities suggest past events of outsiders casting nets over mangrove islands to remove an entire roost of Lilacines at a time. Research has shown that the removal of adults from a population can have more drastic consequences on population size and growth rate, than removal of chicks [40]. In a study of illegal wildlife trade markets in Bolivia, contrary to the idea that most parrots come from nest poaching, 70% of parrots were adults [41]. Our results also suggest that 60% of Lilacines caught in the last three years were from mangroves, so are likely to be adult or juvenile birds. A number of studies have shown that anti-poaching efforts, in the form of additional human presence, can benefit bird populations [42–44] and that recruitment of young people (who may be facilitators in parrot trapping) from the local community to act as nest monitors, can significantly decrease poaching rate [45]. In some cases, nest protection implemented at the correct time of year can have a significant effect [42], but we suggest that year-round protection is needed to safeguard both vulnerable roosting and nesting sites for this species.

The lack of environmental education in Ecuadorian schools is a barrier to reducing parrot ownership and capture [16]. The main purpose of any environmental education strategy is to change people's knowledge and attitudes, and ultimately behaviours [46]. Alone, or in combination with other conservation interventions, environmental education projects can result in a decrease in the persecution of parrots and consequently an increase in population size [43,47]. Most people in our study believed that wild parrots were important for nature and that they themselves had a responsibility to protect parrots. Local people do not want parrots to disappear and are strongly opposed to outsiders coming in to their community to catch them. Contrastingly, up to 74% per community agreed that it was OK to take parrots from the wild to sell, and up to 52% believed that it was OK to keep them as a pet. Furthermore, up to 30% disagreed that catching parrots could make them locally extinct. We found similarity between attitudes and reported behaviours. In areas with more pet Lilacines and reports of parrot trapping, there was also a stronger belief that this was acceptable, compared to areas with fewer pets and trapping. This suggests that changing these attitudes could have an impact on future behaviour, and that the implementation of a targeted behaviour change education project could have conservation benefits to the Lilacine Amazon. We suggest following the practices of the successful PRIDE campaigns [48] which inspire people to take pride in the species and habitats that make their communities so unique, whilst introducing viable alternatives to environmentally destructive practices.

We therefore recommend that a combination of environmental education to change attitudes towards parrot ownership and trapping, and increased protection of wild birds through nest and roost guarding, particularly in the southern part of its range, are conservation priorities for the Lilacine Amazon.

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