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Conservation status and demographics of the Galapagos land iguana (*Conolophus subcristatus*) in the Galapagos Islands, Ecuador

A thesis presented in partial fulfilment of the requirements for the degree of

Masters of Natural Science in Conservation Biology

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New Zealand

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Figure 1.1: A Photograph of a Galapagos land iguana (*Conolophus subcristatus*) roaming on North Seymour Island. Photo: K. Kumar

DECLARATION

I declare that this thesis is an accurate and original account of my research and that the contents have not previously been submitted for a degree at Massey University, nor any other tertiary institution. Except where acknowledged within, the material contained in this thesis has not been written or published by any other individual and to the extent of my knowledge, does not infringe upon copyright restrictions. The research presented here is part of a larger research project on Galapagos land iguanas (*Conolophus subcristatus*) developed since 2012 by my supervisor Dr. Luis Ortiz-Catedral and Alizon Llerena in conjunction with the Directorate Galapagos National Park and Island Conservation. Dr. Luis Ortiz-Catedral allowed me to execute this project following approved protocols by the Ministry of Environment, Ecuador via the Directorate of the Galapagos National Park under permit PC-96-18, see Appendix B on page 18. Dr. Ortiz-Catedral and I coordinated all the logistical support necessary to complete this investigation, and also obtained the relevant permissions for accessing study sites. Dr. Ortiz-Catedral and I jointly prepared funding applications to cover the costs incurred during this investigation. Financial support for the project was obtained from the Mohamed bin Zayed Species Conservation Fund, Galapagos Conservation Trust and Massey University Auckland. The ideas about the components of this thesis were discussed between my supervisor and I prior, during and after fieldtrips. Dr. Ortiz-Catedral provided all the supervision necessary for the development of this project, the analysis of results and the elaboration of the final document. I assume all responsibility for mistakes or omissions present in this document.

Kirtana Kumar

Auckland, New Zealand, January 2019

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ABSTRACT

In conservation biology, understanding the basic demography of populations across their range, provides opportunities to identify and prioritise management actions to improve the conservation of both, populations and of the species as a whole. The Galapagos land iguana or land iguana (*Conolophus subcristatus*) (endemic to the Galapagos Islands), is an ideal model to explore these ideas and to generate information that can assist its conservation. Currently, the species is classified as IUCN: Vulnerable. Land iguana populations are threatened by introduced species and habitat loss but the intensity of these threats varies between populations. The population sizes across the species range are partially known and suspected to relate to the presence/absence of introduced predators. Although demographic trends are virtually unknown, it is suspected that populations in disturbed areas have a lower recruitment of juveniles and limited population growth. Overall, these gaps in knowledge represent a challenge to a formal evaluation of the species' conservation status. Historically, translocation of land iguanas to predator-free or predator-managed sites in the Galapagos Islands has played an important role for its conservation. Since the 1970s there have been calls for translocating land iguanas to Santiago Island, considered to harbour one of the largest land iguana populations until the early 1900s. Although the mechanism for the extinction of the Santiago Island population is not understood, it is most likely linked to habitat changes driven by humans and the effects of introduced predators, which are common denominators in significant iguana population declines on other islands around the world. Since the early 2000s, introduced predators have been eradicated on Santiago Island, and the possibility of reintroducing land iguanas there has resurfaced with greater intensity. While the habitat on Santiago Island is considered suitable for land iguanas, and the expertise to handle and transfer these organisms exists there are missing gaps in knowledge in particular: is the size of historically translocated populations comparable to undisturbed populations? What are the current population sizes of land iguana populations? and what are the effects of introduced predators on land iguana populations?

This thesis addresses these gaps in land iguana demography by providing an update of the current conservation status, estimates of population size and an analysis of the effects of feral cats using historical data. I examined the effects of feral cats on two populations by comparing historical

demographic data from cat-absent and cat-present populations. I found cat-present populations exhibited significantly different population structure and morphology to cat-absent populations, including significantly lower juvenile percentage indicating reduced juvenile recruitment as a result of juvenile depredation . Further, I studied a translocated population of land iguanas (North Seymour) and the most undisturbed land iguana population in one of the most active volcanoes worldwide; Fernandina Island. I estimate and compare population size to assist the planning of a reintroduction of the species to Santiago Island. Finally, I review the IUCN status of the species and conclude that the Vulnerable status is justified, primarily due to its limited geographic range and projected population decline. Four weeks before the completion of this thesis, the Directorate of the Galapagos National Park, Island Conservation and Massey University completed a transfer of over 2000 land iguanas to Santiago Island based on the information provided in this document. Although I could not incorporate the post-translocation monitoring of land iguanas as part of my thesis, I discuss future research needs that will contribute to the restoration of Santiago Island, and the conservation of land iguanas across their range.

RESUMEN

El conocimiento sobre la demografía básica de poblaciones en el ámbito geográfico de especies sujetas a conservación, ofrece oportunidades para identificar y priorizar acciones encaminadas al mejoramiento tanto de las poblaciones como de la especie. La Iguana terrestre de Galápagos (*Conolophus subcristatus*), endémica a las Islas Galápagos, es un modelo ideal para explorar esas ideas y para generar información que pueda asistir en su conservación. Actualmente, la especie esta clasificada como IUCN: Vulnerable. Las poblaciones de Iguana terrestre están amenazadas por los efectos de las especies introducidas y perdida de hábitat, pero la intensidad de esas amenazas varia entre las diferentes poblaciones. Los tamaños poblacionales en el ámbito de la especie son parcialmente conocidos y se sospecha que están relacionados con la presencia/ausencia de depredadores introducidos. Aunque las tendencias demográficas son virtualmente desconocidas, se sospecha que las poblaciones en áreas perturbadas tienen un menor reclutamiento de juveniles y crecimiento poblacional limitado. En general esos vacíos en el conocimiento de la especie representan un reto para la evaluación formal del status de conservación de la iguana terrestre. Históricamente, las reintroducciones de iguanas terrestres a sitios libres de depredadores, o donde estos son manejados han jugado un papel importante en la conservación de la especie. Desde la década de 1970 ha habido sugerencias para la reintroducción de la especie a la Isla Santiago, que se considera tuvo una de las poblaciones de iguana mas grandes en el archipiélago hasta el comienzo de los 1900. Aunque el mecanismo de extinción de la población de iguanas terrestres en Santiago se desconoce, esta muy probablemente ligado a los cambios en el hábitat producidos por humanos y los efectos de las especies depredadoras introducidas, mismos que son los denominadores comunes de reducciones significativas en poblaciones y especies de otras iguanas a nivel mundial. Al inicio de la década de los 2000, depredadores introducidos fueron erradicados en la Isla Santiago, y la posibilidad de reintroducción para la iguana terrestre ha resurgido con mayor intensidad. Mientras que el hábitat en la Isla Santiago se considera adecuado para las iguanas terrestres, y la experiencia para manejar y transferir estos organismos existe, hay vacíos significativos de información. En particular: ¿Es el tamaño de las poblaciones reintroducidas comparable a las poblaciones remanentes no perturbadas? ¿Cuáles son los tamaños poblacionales de las poblaciones de

iguanas terrestres? ¿Cuáles son los efectos de los depredadores introducidos en las poblaciones de iguanas terrestres?

La presente tesis explora esas interrogantes y provee una visión actualizada del status de conservación actual de la iguana terrestre, estimaciones poblacionales y el análisis de los efectos de la presencia de gatos utilizando datos históricos. Examine los efectos de los gatos ferales en dos poblaciones comparando datos demográficos históricos de poblaciones donde habitan gatos ferales y donde no habitan. Determine que en las poblaciones donde coexisten gatos ferales e iguanas terrestres, se verifica una estructura poblacional diferente y también hay diferencias significativas en morfología respecto a poblaciones donde no hay gatos ferales. Específicamente, una proporción menor de juveniles como resultado de la depredación por gatos. Además, estudie una población reintroducida de iguanas terrestres (Seymour Norte) y la población de iguana terrestre menos perturbada en uno de los volcanes mas activos del mundo: Isla Fernandina. Finalmente, revise el status IUCN de la especie y concluyo que el status Vulnerable se justifica principalmente debido al limite geográfico que ocupa la especie y el decremento poblacional estimado. Cuatro semanas antes de completar esta tesis, la Dirección del Parque Nacional Galápagos, Island Conservation y Massey University completaron la reintroducción de mas de 2000 iguanas terrestres a la Isla Santiago, en base a información presentada en este documento. Aunque no pude incorporar datos de monitoreo post-reintroducción como parte de mi tesis, discuto necesidades de investigación a futuro que contribuirán a la restauración de la Isla Santiago y a la conservación de la iguana terrestre en su todo su ámbito geográfico.

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CHAPTER 1

General Introduction



Figure 1.2: A Galapagos land iguana (*Conolophus subcristatus*) on North Seymour Island. Photo: L. Ortiz-Catedral

ABSTRACT

Biodiversity loss is one of the greatest issues affecting the globe. Island species in particular are threatened with extinction due to various anthropogenic factors, in particular the introduction of predators. Several species of iguana (Squamata: Iguanidae) inhabit island ecosystems and many of these represent some of the most threatened reptiles in the world. In order to identify conservation management approaches to restore island populations of iguanas and to reduce their extinction risk, it is important to obtain basic demographic information. The Galapagos land iguana (*Conolophus subcristatus*) (IUCN: Vulnerable) is an ideal model to understand the effects of introduced predators on island ecosystems, as it inhabits several islands across a gradient of accessibility and various stages of ecological integrity; for instance Fernandina Island, the westernmost and youngest of the Galapagos Islands with little human influence to Santa Cruz Island in the middle of the archipelago with a large and growing human population. In this Chapter, I outline threats to island iguanas, common denominators of extinction risk and recovery and present a general description of this thesis.

INTRODUCTION

Human-induced biodiversity loss is a global phenomenon and substantial evidence indicates declines of biodiversity across taxonomic, phylogenetic and functional levels, from the loss of genetic diversity in food crops to the collapse of ecosystem level functions (Lomolino, 1985, 2005; Schmidt & Wei, 2006; Tershy et al., 2016; Valiente-Banuet et al., 2015). There is abundant of evidence showing declines of diversity across taxonomic, phylogenetic and functional levels. For instance, the widely used North American soybean (*Glycine max*) crop has been reduced down to an extremely low level of diversity due to genetic bottlenecks and selective breeding for enhanced crop production, putting the integrity of soybean crop at risk (David et al., 2006). Ecosystem functions collapsing as a result of biodiversity loss is more specifically a consequence of species extinctions or the loss of abundance of certain key species in any given trophic level.

An example of ecosystem function collapse can be seen in some of South America's rivers; the loss of a dominant, migratory, detritivorous fish called the Coporo fish (*Prochilodus mariae*) has altered carbon flow and ecosystem function by decreasing transportation of organic carbon successively downstream. The transport of organic carbon which is aided by *P.mariae* is a fundamental energy source for downstream communities and is also a central biogeochemical function of rivers. Overharvest and dewatering river channels from deforestation have contributed to *P.mariae* decline (Duque, Taphorn, & Winemiller, 1998; Taylor, Flecker, & Hall, 2006). Other examples of biodiversity loss are the particularly high rate of species extinctions that have occurred since the onset of the Anthropocene (Rockstrom et al., 2009). For example, Mora et al. (2011) indicate a dramatic decline of species from a widely agreed previous estimate of 3-100 million to now approximately 8.7 million eukaryotic species (±1.3 million SE), which includes marine and terrestrial species. The estimate is based on existing species databases and taxonomic data.

Species extinctions are a natural phenomenon and prior to the global proliferation of humans, they occurred at a rate of roughly 0.1-1 species per million species per year or lower (Dirzo & Raven, 2003). However, recent estimates show current extinction rates are up to 1000 times higher than the background. An assessment conducted by De Vos et al. (2015) concludes that there is consistent evidence to support a background rate of extinction that is less than one extinction per million species-years. Moreover, De Vos et al. (2015) indicate that future extinctions rates will be as high as 10,000

times the background rate (Barnosky et al., 2011; Harnik et al., 2012; Pimm et al., 2014, 1995). An animal group that currently exhibits this extreme rate of species extinction and endangerment is the amphibians; approximately 34 amphibian extinctions have been recorded since 1500, this does not include the extinction of species that have not been formally identified, if included, the number of extinct amphibian species would likely exceed the initial estimates (Ceballos et al., 2015). Additionally, over a third of known amphibian species are at risk of extinction (McCallum, 2007). Similarly, other animal taxa are also facing extremely high species declines, many off which have gone undocumented (Brook, Sodhi, & Ng, 2003; McKinney, 1999; Thomas et al., 2004).

According to the International Union for Conservation of Nature (IUCN) over 23,000 species are at a high risk of extinction, the IUCN holds the most comprehensive catalogue of threatened species worldwide, where their Global Series Program holds data on over 79,000 species both threatened and non-threatened. Approximately 64,000 of these species are considered as "well documented", this includes information on population size, conservation actions, threats, ecology and utilization (IUCN , 2017). Despite being the most comprehensive list of species worldwide, the number of undescribed species dwarfs the described 79,000 by IUCN. Under 2% of known species are described by the IUCN Red List (Rodrigues et al., 2006). Additionally, many of the species on the list lack a comprehensive or substantial description that is up-to-date, therefore not reflecting the current status of a vast number of species. Some highly data deficient groups, that is, groups for which there is a lack of information on taxonomy, geographic distribution, population status and or threats include reptiles and fishes. (Bland et al., 2015; Derek et al., 2004). Due to these issues, the credibility of the IUCN Red List and whether it should be used as a guide for prioritizing conservation goals for species has been questioned, nevertheless the IUCN Red List continues to be used globally to guide political decisions in species conservation (Campbell, 2012; Ducatez & Shine, 2017; Maes et al., 2015; Mrosovsky, 1997; Rodrigues et al., 2006). Conservation biologists thus have a duty to provide the most up-to-date information for their study species to their local IUCN representatives, in order to maintain an updated global database that truly reflects the conservation status of species.

Conservation goals specifically on those directed towards protecting biodiversity and preventing species extinctions are hindered due to the lack critical information on species and demography. These knowledge gaps need to be addressed if the loss of biodiversity is to be reduced on

a global scale. Besides the consequences at the ecosystem level connected to biodiversity loss, this too can effect human well-being. For example, the removal of apex predators like sharks can have cascading effects on trophic levels in the ecosystem, affecting not only marine communities but also the societies that depend on marine resources, like fisheries. The South-eastern coast of the United States of America scallop fisheries have been out of business since 2004 due to heavy exploitation of sharks, including black tip sharks (*Carcharhinus limbatus*), tiger sharks (*Galeocerdo cuvier),* scalloped hammerheads (*Sphyrna lewini*), and several more. The reduction in shark numbers across the coast have resulted in a bloom of their mesopredatory elasmobranch prey (rays, skates and small sharks), notably the Cownose ray (*Rhinoptera bonasus*). Consequently, the highly abundant cownose ray population consumes sizeable quantities of bivalves, specifically bay scallops (*Argopecten irradians*). Local communities were dependent on North Carolina's bay scallop fishery, however, in 2004 the business had come to an end as the population of bay scallops was so heavily predated on by the cownose ray. Furthermore the increase in in this ray population has greatly reduced the numbers of other bivalves which is now predicted to have potential effects on as far as the lower trophic levels of primary producers such as sea grasses (Myers et al., 2007). This is one of many examples where the loss of biodiversity and its consequences affect human livelihood. In the last two decades there has been a surge in research and literature on understanding how biodiversity loss affects the functioning of ecosystems; supply of goods and service and the health of the planet (Cardinale et al., 2012). The topic generates most interest because ecosystem services provide humanity vital goods and services that human populations rely heavily upon.

Species extinctions have been widely documented in scientific literature in the last two decades, surrounding the discovery of the exceptionally high rate of species extinctions in the 21st century as mentioned earlier (Barnosky et al., 2011; Cardinale et al., 2012; Ceballos et al., 2015; De Vos et al., 2015; Dirzo & Raven, 2003; Pimm et al., 2014). One of the main concerns being that evolutionary responses cannot compensate for the recent and current rate of species extinctions within historical times (Santamaria et al., 2012). Additionally, population level extinctions are three times the rate when compared to the rate at species level, resulting in a significant loss in genetic diversity at the population level (Hughes, Daily, & Ehrlich, 1997). Thus, many researchers have argued that conserving

populations is just as important as conserving the species as a whole , to maintain ecosystem functions and conserve biodiversity (Derek et al., 2004; Frankham, 1995, 2010).

SPECIES EXTINCTIONS ACROSS TAXONOMIC GROUPS

Extinction rates vary widely across taxonomic groups. Generally, the scientific data and records of extinctions and threatened species from the past few centuries have been focused on mammals and birds. For instance, McKinney (2001) suggests that mammalian species extinction and threat estimates represent the most accurate indicators of the real rate of threat and extinction because they are among the most heavily studied taxa. Furthermore, McKinney (2001) reports major underestimates of species extinctions in poorly studied taxa groups such as insects and marine invertebrates, suggesting that their true extinction rates are not accurately represented in the literature. Similarly another study shows that 31% of scientific papers published in 2001 in three renowned conservation journals are focused on birds and mammals and conclude that these taxa are overrepresented within vertebrate studies (Titley, Snaddon, & Turner, 2017).

Thus, it is apparent that there is bias in the literature and in conservation attention that favours birds and mammals. There is also an evident bias in number of evaluated species buy the IUCN, with almost all extant birds and mammals being formally assessed as opposed to other vertebrate groups such as amphibians and reptiles with only 41% and 45% of species assessed respectively (de los Ríos, Watson, & Butt, 2018). It is only at the beginning of the 20th century that reptiles, amphibians and freshwater fish extinctions have received greater attention. Highly data deficient groups include reptiles and fishes: data deficient meaning there is a lack of information on taxonomy, geographic distribution, population status and or threats (Bland et al., 2015). The lack of formal evaluation and research attention highlights the increased risk of extinction for these poorly studied taxa groups.

Bland & Böhm (2016) investigate the issue of data deficiency in reptiles. Their research shows that it is still common for many reptile species to be discovered every year. For example in 2008 and 2009 42 new species of reptiles were discovered (Ceballos, García, & Ehrlich, 2010). These discoveries were restricted to Southeast Asia and Eastern Himalayas In 2016, 11 new reptile species were discovered in just the Greater Mekong region alone (WWF, 2016). This suggests there could be well over 10 new reptile discoveries every year globally. Thus there is a lag between documenting reptile

diversity and assessing their conservation status in order to update the IUCN Red List. Bland & Böhm (2016) also mention how reptiles as a group are a poorly represented taxon in the IUCN Red List of Threatened Species. Böhm et al. (2013) attempt to evaluate the extinction risk of reptiles for the first time. A randomly selected sample size of 1500 reptiles worldwide was used in the extinction risk assessment. The assessment concluded that the most threatened families within the reptilian taxon include: *Geoemydidae* and *Chelidae* (Turtles), *Crocodylidae* (Crocodiles), *Pygopodidae*, Xantusiidae and *Iguanidae* (Lizards). One in five reptile species are threatened with extinction, with the highest risk of species within freshwater habitats and tropical regions and one in five lizard species are also threatened with extinction according to this study (Böhm et al., 2013). 318 of the 1500 reptiles assessed were data deficient, consequently they were not assigned a level of extinction risk, instead it was indicated that further information will need to be collected in order for them to be assigned to a risk category.

The data deficient species causes an uncertainty in extinction risk patterns (Bland & Böhm, 2016). Moreover, data deficient species are rarely included in national recovery plans and conservation planning. Conservation funds are restricted to species with high extinction risk, which disfavours poorly known taxa (Howard & Bickford, 2014). An example can be shown with the Mohamed bin Zayed Species Conservation Fund (MBZ) awarding funds to only 2% of data deficient species (Bland & Böhm, 2016). It is crucial for data deficient species to be examined to alleviate the uncertainty surrounding their risk of extinction, as well as addressing the data gaps in scientific knowledge of biodiversity indicator species and to assist in the conservation of species with adequate information. A study by Bland et al. (2015) also similarly concludes that data deficient species are likely to go extinct before any action is taken unless they are specifically targeted for monitoring. However, in a more recent article, it is suggested that often research and conservation efforts are generally prioritized and directed towards data deficient species and species with higher threat categories such a as 'Critically Endangered' (Járic et al., 2017). That leaves species under the lower threat categories of 'Near Threatened' and 'Vulnerable' to perhaps receive decreased research attention and the need for urgency in conservation action, affecting their overall risk of extinction.

REPTILE DIVERSITY AND THREATS

Reptiles are the second most species-rich group within vertebrates after birds and are widespread on a global scale. The array of habitat types they occupy has enabled them to acquire a vast diversity of, behavioural, ecological, morphological, life history, and defensive strategies in the face of any environmental event or demand (Pincheira-Donoso et al., 2013). Extant reptiles are those that have evolved since the extinction event at the end of the Mesozoic era. The greatest diversification can be seen specifically in snakes and lizards (Hickman et al., 2015). Lizards and snakes belong to *Squamata*. *Squamata* is the most diverse and successful lineage with over 9,400 species (Pyron, Burbrink, & Wiens, 2013). This reflects over 95% of extant reptile species. The second most diverse is *Chelonia* which includes all turtles with 327 species, followed by *Crocodila* with 25 species of crocodile and *Rhyncocepalia* with only one species, the Tuatara (Pincheira-Donoso et al., 2013). Some species within *Squamata* have been able to radiate and occupy a range of climates with colder temperatures unlike crocodiles and turtles. There are over 5,800 species of lizards (de la Navarre, 2013) making up the majority of Squamate species, they include terrestrial, burrowing, aquatic, arboreal and aerial individuals (Hickman et al., 2015). A highly diverse group is iguanian lizards, with nearly 400 known species within the *Anolis* genera and over 220 species within the *Liolaemus* genera (Pincheira-Donoso et al., 2013).

The extinction risk of reptiles is potentially inflated due to their data deficient nature. Reptiles in different geographic locations face different threats. Reptiles on islands in particular are at an even greater risk of extinction, due to the high extinction rate of species on islands paired with the shortfall of information on reptiles in general (Bland & Böhm, 2016; Foufopoulos, Kilpatrick, & Ives, 2011; Frankham, 1998). It is well documented that species extinctions in modern times are considerably higher on islands than when compared with mainland extinctions, literature also shows species becoming endangered between 1985 and 1995 have mostly occurred on islands (*Science and the Endangered Species Act*, 1995). A study of extinction rates in 1993 show a much higher number of recorded animal extinctions on islands with 367 extinctions, as opposed to 124 animal extinctions on the mainland (Smith et al., 1993). (Frankham, 1998) indicates that 75% of animal extinctions worldwide have occurred on islands since the 1600s. Being small and isolated, it is no surprise that the rate of extinctions on islands exceeds those recorded on the mainland; only smaller populations can inhabit

islands due to habitat availability constraints, making island inhabitants vulnerable to extinction. A number of other factors contribute to high risk of extinction on islands including susceptibility to environmental stochasticity with sudden occurrence of disease or natural disaster, easily wiping out an entire population, inbreeding depression is also argued to be another contributing factor. Moreover island endemic species tend to have higher extinction rates than non-endemic species (Frankham, 1998). As for reptiles, the majority of extinctions in the last 10,000 years have occurred on islands. A study by Case et al. (1992) concluded that previous extinctions of reptiles have all been due to human-related disturbances. Additionally, they concluded human-related extinctions were increased for islands of smaller sizes. In both cases, the authors identified the chief cause as the introduction of predators. This is corroborated with other research on island extinctions (Li et al., 2014; Manuel et al., 2004; Medina et al., 2011) . Approximately 37 reptiles have been classified as extinct since the 1500s and the majority of these extinctions (24) have occurred since the 1900s, this is a conservative estimate based on the IUCN Red List (Ceballos et al., 2015).

APPROACHES TO HALT SPECIES EXTINCTIONS ON ISLANDS: PAST REPTILE CONSERVATION CASE STUDIES

Translocations

Conservation biology has evolved in the last few decades into an increasingly multifaceted science (Hall & Fleishman, 2010). Species translocations as a conservation tool has consequently only occurred just over a century ago, however most have only occurred in the last 70 years following the birth of modern conservation biology (Atkinson, 2001). Translocations are a sub-discipline within restoration ecology and can be the reintroduction of species where an animal is released into an area that was part of its historical range, but is now locally extinct from that certain area; or an introduction whereby the animal is translocated to an area out of its historical range (IUCN/SSC, 2013; Seddon, 2010). Translocations have been used widely to save species from extinction. Birds in particular have been heavily translocated in recent history. From the years 1973 to 1986, 93 species were translocated within Australia, New Zealand, Canada and the United States. 59% of the translocations were birds, 38% mammals and 3% were reptile translocations (Griffith et al., 1993). The bias towards mammals and birds in species translocations remains the same today, often regardless of their degree of threat or vulnerability (Seddon et al., 2014). Successful translocation efforts and island restoration is especially

renowned in New Zealand. The introduction of mammals following Polynesian and European settlement particularly since the 1800s had caused severe declines in New Zealand native species, namely native birds, of which 27 species have gone extinct following human settlement (Bellingham et al., 2010). This rapid loss of native bird species through predation prompted conservationists to adopt translocations and is now a well-established conservation technique in New Zealand. By 1891 some of the offshore islands were established as reserves for the purpose of conserving mainland native bird species in a sanctuary setting. These island sanctuaries were a part of the first species translocation practices in the world. It is common for translocations to be coupled with invasive species eradications. For example the translocations of Saddleback (*Philesturnus*), Brown Kiwi (*Apteryx mantelli*) and Takahe (*Porphyrio hochstetteri),* were accompanied by the eradication of introduced mammals including cats (*Felis catus*) and possums (*Trichosurus Vulpecula*) on several offshore islands of New Zealand (Atkinson & Cameron, 1993; Bellingham et al., 2010).

Despite these past successful translocations in New Zealand, the global success of translocations is regarded is much more modest. The success rate of translocations whereby the reintroduction of the species is well established in a self-sustaining population, is very low at 23%. To deem a population as 'successful' after the completion of a translocation, requires long-term monitoring to see if whether the population is self-sustaining. Duration of post-release monitoring will be dependent upon the generation time of the species, their numbers and area of release site. Initial establishment, successful reproduction and long term persistence are all indicators of a successful translocation (Tarszisz, Dickman, & Munn, 2014). A reason for such a high failure rate in translocations could be due to the lack of long-term monitoring. Many post-release monitoring operations are not long enough; they generally range from one month to five years. The amount of funds available in the project is an important factor in determining the duration of post-release operations (Tarszisz, Dickman, & Munn, 2014). Many authors stress the importance of adequate post-release monitoring for achieving successful translocations (Armstrong et al., 2015; Batson et al., 2015; Parker et al., 2013; Seddon, Armstrong, & Maloney, 2007).

In recent years reptile translocations have proven to be successful more often than not, with post-release monitoring schemes in place to assess translocation and establishment success (Bell & Herbert, 2017). An excellent example of successful translocations can be seen with iguana species from

the Bahamas and Galapagos archipelagos. Viable populations have been established post-release in many iguana species, most within the *Cyclura* genus such as the Allen Cays iguana (*C. cychlura inornata*), Exuma Island iguana (*C. cychlura figginsi*), Acklins iguana (*C. rileyi nuchalis*), Anegada iguana (*Cyclura pinguis*), Cuban iguana (*C. nubila nubile*) and also land iguanas (*Conolophus subcristatus*) (Alberts, Carter, & Hayes, 2004). Despite their general prevalence in nature and the extreme diversity of reptile groups, they are largely underrepresented in species translocations. Reptile species in New Zealand are one of the few that are known to have a long history of translocations. Species such as the Tuatara (*Sphenodon spp.*), a number of gecko species and skink species have been translocated numerous times since the 1980s. Similarly like many other reptile species that occur on islands, New Zealand reptile species are prone to extinction with the presence of introduced mammalian predators, which coupled with the insular nature of island populations has contributed to historical losses (Towns et al., 2016). Again, conservation efforts included a combination of translocations (reintroductions) and mammalian eradications to minimize extinction risk for these species, which is an indication of the importance of a multifaceted approach to species conservation.

Invasive species management and predator control

Invasive mammals are the largest threat to island biodiversity, causing the majority of extinctions in the last two centuries (Anderson et al., 2016; Jones et al., 2016). As mentioned earlier the coupling of translocations with invasive species control is a powerful conservation tool for mitigating native or endemic species decline and extinctions. New Zealand is the first country to have developed methods used today to manage invasive species and has been well established for over three decades (Howald et al., 2007). Up until the early Cretaceous, New Zealand historically had no terrestrial mammalian species (Worthy et al., 2006); the only native mammals are a small group of bat species from the *Mystacinidae* and *Vespertilionidae* family. The prime group of native animals residing in New Zealand are birds and then reptiles (Barnett, 1985). A large number of plant and animal species had been introduced since human settlement (Pech & Maitland, 2016). Introduced mammals, in particular mustelids and rodents have been by far the most harmful to New Zealand species causing many extinctions and population declines of natives species across the country (Russell et al., 2015). Additionally, endemic islands species are most vulnerable to introduced mammalian predators

(Frankham, 1998) because they have not evolved behavioural defences against mammalian predators, thus can suffer severe population reductions (Medina et al., 2011). Control of invasive species is paramount in conservation management. Methods used to control these animals include trapping or using ground-based or aerial poisoning with a variety of poisons including sodium monofluroacetae (1080), brodifacoum and cyanide. Today introduced mammals have been successfully eradicated from over 100 offshore islands (Russell et al., 2015). As is the case in New Zealand, feral cats are one of the most harmful introduced mammals to native and endemic species worldwide. Feral cats have contributed to at least 14% of reptile extinctions and 8% of reptile declines (Manuel et al., 2004; Medina et al., 2011). Eradication methods developed in New Zealand have been largely influential in other parts of the world (Howald et al., 2007) such as the Galapagos islands where introduced rodents and feral cats are among some of the introduced mammals that threaten a large number of native and endemic species including Galapagos land iguanas.

The majority of reptile population declines and recent extinctions are well exemplified by Iguanas, many of which inhabited the West Indian and Fijian archipelago (Manuel et al., 2004; Medina et al., 2011). Much of the literature on predator control is based on islands, invasive species have accounted for half of island extinctions (Russell et al., 2015). Despite this high level of invasive predator-driven island extinctions, many successful eradications have aided a number of acutely threatened species. Iguana species in the Galapagos and West indies in particular have benefitted from cat eradications and translocations (see case studies below).

Large-scale habitat restoration

Restoration ecology is a relatively new science that has developed in the last three decades (Young, 2000; Andel & Aronson, 2012). Restoration ecology goals have over the years moved from the traditional view of approaching an equilibrium in a static environment, to a more non-equilibrium and dynamic view with the realisation of major and accelerating environmental changes such as climate change and biodiversity loss. Thus, contemporary restoration goals have been modified from previous to adjust and account for the sudden environmental changes. Instead, the goal of resilient ecosystems is an advance of the traditional view of trying to revert back to the original state of the environment (Hilderbrand, Watts, & Randle, 2005). Environmental resilience being the capability of withstanding

disturbances and absorbing any changes an ecosystem may experience, in short the capability of resisting degradation and recovering from disturbance (Perring et al., 2015).

An excellent example of a successful large-scale restoration project can be seen on Santiago Island, in the Galapagos Archipelago. Santiago Island (58,465 ha), used to have populations of feral dogs (*Canis familiaris*), pigs (*Sus scrofa*), goats (*Capra hircus*) and donkeys (*Equus africanus asinus*), all of which were eradicated as part of one the world's largest mammalian eradication projects: Project Isabela (Cruz et al., 2009). Generally mammalian eradications are more feasible on islands of smaller size (Medina et al., 2011). Hence eradicating all large mammals from Santiago Island represents a major milestone in ecological restoration worldwide. The only introduced mammal that is left on Santiago Island are black rats (*Rattus rattus*), these have proven to be difficult to remove as it is complicated to target just the black rat species when there the endemic rice rat (*Nesoryzomys swarthi*) is also present on the island (Harris & Macdonald, 2007).

IGUANAS

Iguanas (*Iguanidae*) are part of one of the most highly diverse groups; Reptilia. There are eight extant genera which include 44 iguana species in *Iguanidae*, the most commonly accepted taxonomy (Buckley et al., 2016; Buckley et al., 2011). Under this view, the genera are: *Cyclura*, *Ctenosaura*, *Amblyrynchus*, *Conolophus*, *Sauromalus*, *Iguana*, *Brachylophus* and *Dipsosaurus* (Etheridge & de Queiroz, 1988; Malone, Reynoso, & Buckley, 2016, 2017).

However, Alberts, Carter, & Hayes (2004) state that there are as many as 11 genera iguanid genera. The taxonomy of *Iguanidae* is therefore controversial as there are substantial differences in the number of genera and species recognised between authors (Schulte, Valladares, & Larson, 2003). Taxonomic revisions are ongoing and result in frequent reclassifications and discoveries of new species or subspecies. As methods for phylogenetic and taxonomic studies advance and the availability of evidence widens, new findings are published. For instance taxonomic revisions for marine iguanas in 2017 have uncovered 11 subspecies within the *Amblyrynchus* genus, from the previously known 7 subspecies (Miralles et al., 2017). In 2009 the Galapagos Pink iguana was recognized as a separate species (*Conolophus marthae*), while before it was considered as a colour morph of the common yellow land iguana (*Conolophus subcristatus*) (Gentile et al., 2009).

Iguanas are large herbivorous lizards that occupy a diverse range of habitats and exhibit a varied range of social, behavioural and morphological characteristics across species (Alberts, Carter, & Hayes, 2004). The majority of iguanas reside in the Americas and surrounding oceanic islands. Fewer species occupy the Fijian and Tongan islands, in the South Pacific. High levels of endemism are a characteristic of island iguanas. Five out of the eight iguana genera exhibit high or total endemism (Alberts, Carter, & Hayes, 2004; Palmeirim, 2007; Rassmann et al., 1997; Zarza, Reynoso, & Emerson, 2011). Highest levels of diversity and number of species are found in the *Cyclura* and *Ctenosaura* genera, which make up over half of all iguanid species (Buckley et al., 2016).

Cyclura iguanas, commonly known as rock iguanas, are a terrestrial group, which occur across the dry rocky islands of the West Indies. Many of the species and sub species from this genus are restricted to a single island, which is why they have the highest level of endemism within iguanids (Alberts, Carter, & Hayes, 2004; Malone et al., 2000). *Ctenosaura* iguanas are characterized by spiny scales on the tail, which give them the common name of spiny-tailed iguanas, they are native to Central America and Mexico and are found terrestrially in dry forests (Alberts, Carter, & Hayes, 2004). 18 different species belong to this genus (Buckley et al., 2016). *Conolophus* and *Amblyrynchus* iguanas are endemic to the Galapagos Islands. *Conolophus spp.* are known as Galapagos land iguanas, with three different terrestrial species occurring in the drier rocky regions of the islands. *Amblyrynchus* is the only genus containing species of marine iguana and has a wide distribution within the Galapagos Islands, with several subspecies that occur in large numbers along the coast of different islands in the archipelago (Rassmann et al., 1997). *Sauromalus* (commonly known as Chuckwalla's) and *Dipsosaurus* are known as desert iguanas, they are found terrestrially on the mainland and on some islands of the Southwestern United States and Mexico (Lamb, Jones & Avise, 1992). *Iguana* and *Brachylophus* are arboreal genera, which occur in wet forests. *Iguana* is a common and widespread genus that is native to Mexico, Central and South America, as well as the Caribbean. They have also been introduced and are invasive species in parts of Florida, a small part of Japan, on other Caribbean islands and on two Hawaiian islands: Maui and Oahu. As a genus, *Iguana* has been the subject of many behavioural studies and was the first of the iguana species to be thoroughly studied, they are kept commonly as pets (Falcón et al., 2013). In contrast to all other iguana genera, *Brachylophus* is the only iguana genera that resides in the Old World region on the islands of Fiji (Alberts, Carter, & Hayes, 2004).

Due to the majority of iguana species and populations occurring on islands, they are particularly susceptible to extinction and population declines. Despite even having high population densities, demographic and environmental stochasticity, restricted and reduced land area, dispersal barriers and their insular adaptations increase their vulnerability to extinctions at large when compared to mainland populations. 75% of recorded animal extinctions since 1600 have been island dwelling species; this percentage includes all of the 28 known reptile species extinctions (Alberts, Carter, & Hayes, 2004; Frankham, 1998; Honegger, 1981; Wood et al., 2017). The introduction of alien predators to islands has caused the most destructive and rapid examples of local species extinctions. 86% of vertebrate island extinctions since 1500 A.D. have been attributed to invasive species (Spatz et al., 2017). The extinction of a population of Turks and Caicos rock iguana (*Cyclura carinata)* in Pine Cay Island is a prime example of a local extinction caused by introduced predators. An estimated 15,000 individuals of this species were extirpated in less than five years, due to the introduction of feral cats and dogs (Harlow & Biciloa, 2001). Iguanid lizards require larger undisturbed areas unlike smaller lizard species such as geckos which thrive in small and altered habitats like urban environments. Being insular large lizards, iguanas are amongst the most endangered lizard species world-wide (Alberts, Carter, & Hayes, 2004).

At present the IUCN Red List has assigned threat categories for iguanas under all eight genera. 35 species are included in this list (IUCN Red List, 2017). Some of them have information on whether their population is increasing, decreasing or stable. Table 1.1 includes 35 iguanas and their given threat category and current population trends assigned by the IUCN. That leaves nine species of iguana that are yet to be assigned a threat category by the IUCN Red List. 54% of the iguana species are in the Endangered of Critically Endangered category and 43% show decreasing population trends. 91% of these species occur on islands. This highlights the vulnerability to extinction for Iguanid species.

Table 1.1: Summary of all species of Iguana in the IUCN Red List, with their associated threat status and current population trends. Information sourced from: https://www.iucnredlist.org/. IUCN Threat Status Key: (NT= Not Threatened; LC= Least Concern; VU= Vulnerable; EN= Endangered; CR= Critically Endangered; DD= Data Deficient)

Genus	Species	Common Name	Threat Status	Current Trend
Amblyrhynchus	*cristatus*	Fernandina Marine Iguana	VU	Unknown
Brachylophus	*bulabula*	Fiji Banded Iguana	EN	Decreasing
	fasciatus	Lau Banded Iguana	EN	Decreasing
	vitiensis	Fiji Crested Iguana	CR	Decreasing
Conolophus	*marthae*	Galapagos Pink Land Iguana	CR	Unknown
	pallidus	Santa Fe Land Iguana	VU	Unknown
	subcristatus	Common Land Iguana	VU	Unknown
Ctenosaura	*alfredschmidti*	Campeche Spiny-tailed Iguana	NT	Unknown
	bakeri	Utila Spiny-tailed Iguana	CR	Decreasing
	clarki	Balsas Spiny-tailed Iguana	VU	Unknown
	defensor	Yucatán Spiny-tailed Iguana	VU	Unknown
	flavidorsalis	Yellow-backed Spiny-tailed Iguana	EN	Decreasing
	melanosterna	Black-chested Spiny-tailed Iguana	EN	Decreasing
	nolascensis	Nolasco Spiny-tailed Iguana	VU	Stable
	oaxacana	Oaxaca Spiny-tailed Iguana	CR	Decreasing
	oedirhina	Roatan Spiny-tailed Iguana	EN	Decreasing
	palearis	Motagua Spiny-tailed Iguana	EN	Decreasing
	praeocularis	Southern Honduran Spiny-tailed Iguana	DD	Unknown
	quinquecariniata	Five-keeled Spiny-tailed Iguana	EN	Decreasing
	similis	Common Spiny-tailed Iguana	LC	Stable
Cyclura	*carnita*	Turks and Caicos Rock Iguana	CR	Decreasing
	collie	Jamaican Iguana	CR	Unknown
	cornuta	Hispaniolan Rhinoceros Iguana	VU	Decreasing
	cychlura	Northern Bahamian rock iguana	VU	Unknown
	lewisi	Grand Cayman Blue Iguana	EN	Increasing
	nubila	Clouded Rock Iguana	VU	Unknown
	pinguis	Cyclura pinguis	CR	Unknown
	ricordii	Ricord's Rock Iguana	CR	Decreasing
	rileyi	Central Bahamian Rock Iguana	EN	Unknown
	stejnegeri	Mona Rhinoceros Iguana	EN	Decreasing
Dipsosaurus	*dorsalis*	Common Desert Iguana	LC	Stable
Iguana	*delicatissima*	Lesser Antillean Iguana	CR	Decreasing
	iguana	Common Green Iguana	LC	Unknown
Sauromalus	*ater*	Common Chuckwalla	LC	Stable
	hispidus	Spiny Chuckwalla	NT	Unknown

APPROACHES TO IGUANA CONSERVATION

Prior to the 1980s iguana research, particularly on green iguanas (*Iguana iguana*) was generally focused on behaviour and basic biology, For instance evolutionary biologist and herpetologist, Dr. Austin Stanley Rand studied the nesting behaviour of green iguanas on an islet in Panama, this was among some of the first behavioural studies of wild green iguana populations. Green iguanas dominated the research for a decade, it was only in the 1970s that other species from different genera were explored in research (Alberts, Carter, & Hayes, 2004). The topic of conservation of iguanas has received more interest in recent decades with the increasing concern for the rapidly changing habitats of insular species on islands and population declines of island iguanas. The destructive effects of introduced mammals on island iguana populations had initiated conservation research and action primarily between the 1970s and 1990s. Iguana species that have been involved in early conservation operations include Galapagos land iguanas, Jamaican Iguanas (*Cyclura collie*), Allen's Cay iguanas (*Cyclura cychlura* inornata) and Anegada rock iguanas (*Cyclura pinguis*). All of these iguanas were threatened with extinction due to heavy predation from introduced mammals (Fabiani et al., 2011; Goodyear & Lazell, 1994; Knapp, 2001; Wilson, Robinson, & Vogel, 2004).

It is well established that insular and endemic species are at a higher risk of extinction (Frankham, 1998). Iguanid lizards are no exception, with 61% of the extant iguana species at risk of extinction according to (Tershy et al., 2016). Declines of iguana population across their genera have been well-documented in the past and research examining conservation measures has been initiated following the decline of populations of iguana species, particularly in the Caribbean and Galapagos islands. As mentioned earlier many of these studies that have been published on an iguanid species are centred on the impact of predation and predator control on islands, as introduced predators are identified as a leading cause in iguanid lizard declines across their ranges (Alberts, Carter, & Hayes, 2004; Tershy et al., 2016). Very few have placed a focus on population demographics. The shortage of historical and demographic data has been shown to negatively impact the management of declining species (Freake & DePerno, 2017). Therefore, conducting baseline demographic studies forms a crucial and necessary component of effectively managing species under conservation management scenarios. Population

demographics can give important insights into the particulars of an iguana population and subsequently appropriate conservation measures can be applied to manage population-specific threats.

Conservation of iguanas has centred on a combination of translocations, breeding programs and invasive species eradications. In particular the use of translocations has played a major role in island iguana conservation, due to their limited ability to disperse outside of their geographic range over ecological timescales (Alberts, Carter, & Hayes, 2004). Translocations of iguanas have occurred as early as the 1930s on the Galapagos Islands (Perkins, 1930). Only in the 1970s did translocations occur solely for the purpose of conservation. Captive breeding programs and captive-breeding centres were established by the 1970s for the Galapagos land iguana. Eradication campaigns were also undertaken to remove feral dogs, pigs and goats in order to save populations of Galapagos land iguanas from extinction (Tzika et al., 2008).

Much of the literature on iguanid lizard conservation, is focused on the control of introduced predators and competitors such as feral cats (*Felis catus*), dogs, black rats and grazing herbivores that either prey on juvenile iguanas or degrade their habitat by grazing the surrounding vegetation (Carrion, 2008, 2011; Harper, 2011). Therefore it is common for translocations to be coupled with intensive eradication programs. The published evidence indicates that the predominant threat to island iguanas is habitat loss along with introduced mammals, (Barnett & Rudd, 1983; Barnett, 1985; Barun et al., 2011; Mitchell, 1999; Mitchell et al., 2002; Phillips et al., 2005). Many of the studies recognise feral cats are one of the most serious threats to iguana populations. Cats prey on juveniles and eggs, thus the lack in juvenile recruitment could have significant negative consequences for population growth and haplotype diversity within populations. There are numerous examples of the negative effects of cat predation specifically on rock iguana populations from the West Indies, which includes all *Cyclura* iguanas (Iverson, 1978, 2007; Mitchell et al., 2002; Nogales et al., 2013). Habitat loss, agriculture and development of tourist resorts is another major threat for West Indian Rock Iguanas (Alberts, Carter, & Hayes, 2004; Knapp & Owens, 2005; Tole, 2002; van den Burg et al., 2018). The eradication of feral cats and rats is a main conservation goal for West Indian Rock Iguana species. A campaign aimed at eradicating feral cats with sodium monofluoroacetate (1080) was put forward in 1999 on Long Cay Island in the Bahamas. Eradication was successful and Turks and Caicos rock iguanas (*Cyclura carinata*) were successfully translocated to Long Cay from Big Ambergris (Mitchell et al., 2002). The

iguana population in Big Ambergris Cay had been severely reduced following the construction of a hotel and the introduction of free-ranging cats and dogs that came with it in 1970s. The introduction of these animals had proven to be the most serious threat to the iguanas here, thus a translocation and eradication was planned to mitigate the effects that these introduced predators had on the endemic iguanas. The population of iguanas on Long Cay are now thriving in the absence of predators of the protected environment to which they were translocated (Mitchell et al., 2002). Habitat loss and the introduction of mammals have also caused local extinctions and major population declines in the iguana populations of the Fijian and Galapagos archipelagos. The historical range of the Endemic Fijian Crested iguana (*Brachylophus vitiensis*) has been reduced by almost 80% during the last three decades, primarily due to habitat loss with forest fires and the introduction of goats which over graze the vegetation that is vital for sustaining the local iguana population (Morrison et al., 2009). Predation by introduced cats and dogs is also a contributing factor. The remaining population of the critically endangered Fijian Crested iguana is now restricted to two islands and although these populations appear to be thriving, they are still highly vulnerable to extinction, again, due to the insularity caused by their limited island distribution (Morrison et al., 2009).

Similarly, other iguana populations such as the marine iguana and land iguanas of the Galapagos Archipelago have experienced ongoing population declines due to the predation of juveniles by feral cats and feral dogs (Kruuk & Snell., 1981, Phillips et al., 2005) . Previous populations of land iguanas have been completely decimated due to the introduction of mammal predators (Cruz et al., 2005). There have been ongoing cat eradication campaigns in the Galapagos and from 2001 to 2004, sodium monofluroacetate (1080) was used to poison feral cats on Baltra Island (Phillips et al., 2005). During this time, approximately 90% of the cat population was eliminated and there was a subsequent increase in the proportion of juvenile iguanas to adult iguanas following the campaign (Phillips et al., 2005).

HISTORY OF GALAPAGOS LAND IGUANA CONSERVATION

Up until the 1960s, the status of land iguana populations in the Galapagos was considered stable, with little risk of extinction (Cayot, 2008). However, since the arrival of early whalers and settlers in the 17th century, all but one island, out of the six islands which hold land iguana populations have faced severe anthropogenic disturbances. Fernandina is the only undisturbed island with a population of land iguanas (Tzika et al., 2008). Many of the populations have been severely reduced in size and some were driven to extinction, such as the populations of Santiago Island (Phillips et al., 2005). The cause of extinction of iguanas from Rabida islet is unknown (Steadman & Zousmer, 1988). Introduction of mammals to Santiago Island had caused the extirpation of a large land iguana population and caused major population declines of iguanas on Santa Cruz and Isabela Island (Cruz et al., 2005). Goats, pigs, rats, feral cats and feral dogs threaten land iguana populations due to competition, habitat destruction and predation of juveniles, adults and eggs (Tzika et al., 2008). In particular, cats pose a major threat as they prey on juvenile land iguanas up to 2 years old (Harper et al., 2011; Marquez et al., 2010). The lack of juvenile recruitment that results from this predation is one of the major factors limiting the success of the species (Phillips et al., 2005). In 1975, the status of land iguana populations had to be reconsidered after several feral dog attacks had reduced the Santa Cruz Island population of land iguanas by over 90% and local populations of iguanas from parts of Isabela Island (Tzika et al., 2008).

Initial conservation efforts to save land iguanas were centred around translocations and breeding programmes which began in the 1970s, eradications of introduced mammals followed later (Cayot, 2008). Efforts to restore land iguana populations on Isabela and Santa Cruz Island began with a captive breeding program initiated by the Charles Darwin Research Station (CDRS) based in Santa Cruz Island. Following the feral dog attacks the remaining population of land iguanas on Santa Cruz consisted of only 56 individuals, all of which were transferred to the station to be initiated into the breeding program. In 1977, 38 individuals from the breeding centre were translocated to Venecia, an islet off the coast of Santa Cruz. It is confirmed that the iguanas that were translocated to Venecia have become well established and are successfully reproducing.

In addition to the conservation efforts on Santa Cruz and Venecia a separate breeding program was established on Isabela Island to help recover land iguana populations there. Additional measures

were taken in 1982 to eradicate the feral dog population and was soon followed by the release of 390 captive-bred individuals back to Cartago Bay, Isabela where they were recently extinct at the local level. Before the 1970s, around 70 iguanas from Baltra Island were transferred for experimental reasons to North Seymour Island, this occurred between 1932 and 1933 (Cayot & Menoscal, 1992; Cayot, 2008). Historically there were no iguanas on North Seymour before this translocation. Ultimately this translocation ensured the survival of Baltra Island iguanas, as shortly after, the land iguana population was wiped out during the construction of the military airbase on Baltra Island. In the early 1990s, North Seymour iguanas were reintroduced to Baltra Island as part of a repatriation program and are now well-established. The individuals released during this translocation represent only a proportion of over 700 captive-bred iguanas that have been repatriated on three different islands. The populations re-established from these translocations are reported to be reproducing successfully on all three islands (Barnett & Rudd., 1983: Cayot & Menoscal, 1992; Cayot, 2008).

These examples provide evidence of the past benefits that captive breeding programmes coupled with translocations and the eradication of invasive mammalian predators has had for the land iguana populations present on the islands of Galapagos. Currently there is no captive breeding programme for land iguanas, this aspect of their conservation was discontinued in 2008 (Cayot, 2008). Efforts are now aimed solely at eradication of invasive mammals, however this not specifically directed towards the conservation of land iguanas (Cayot, 2008). There have been several successful eradications of introduced mammals on a number of islands in the Galapagos (Carrion et al., 2011; Cruz et al., 2005, 2009; Harper, Zabala & Carrion 2011, Phillips et al., 2005). However many of the islands still have large populations of introduced mammals that threaten native species (Harris, 2007;Wiedenfeld, 2008; Harper, 201; Steadman, 1991; Eckhardt, 1972). The eradication of large herbivorous mammals such as goats and pigs has proven to be straightforward compared to the removal of smaller species, where feral cats and rats have proven to be more difficult to eradicate and have only recently been targeted effectively. For instance Baltra had been part of cat eradication campaign with the use of 1080 poison in the early 2000s and North Seymour had been part of the first long-term rat eradication campaign using brodifacoum ($C3_1H_{23}BrO_3$) poison to eradicate black rats in 2007 (Carrion, Sevilla, & Tapia, 2008). Monitoring of the iguana population was not extended after the rat eradication program,

consequently the full benefits of the rat eradication program is unknown (Harper, Zabala, & Carrion, 2011).

There are 11 known main populations of land iguanas (Marquez et al., 2010). This is based on geographical separation, see Figure 1.3. Six of these populations experience juvenile predation by feral cats, thus limiting the juvenile recruitment of these populations of land iguana. The islands with juvenile predation include the islands of Isabela and Santa Cruz (Tzika et al., 2008). Currently, there are no specific conservation measures in place for the Galapagos land iguana. However, the Galapagos National Park Services (GNPS) is interested in translocating land iguanas from North Seymour Island to Santiago Island in the near future. These translocations are for the purpose of conserving the various populations of land iguana, as well as for ecosystem restoration on Santiago Island. Santiago is an ideal site for reintroduction of land iguanas as the successful eradication of large introduced mammals and cats has reduced the likelihood of predation and offers suitable habitat for land iguanas to once again inhabit, with limited disturbance from browsing species.



Figure 1.3: A map of the Galapagos islands showing 11 known extant populations of land iguanas. Red dots indicate those exposed to cat predation, blue dots indicate populations without cats. Modified from \ E. Gaba distributed under CC-BY 2.0 License

ECOLOGICAL RESTORATION OF SANTIAGO ISLAND

Santiago Island in the Galapagos archipelago is an example of a large-scale restoration site (Cruz et al., 2009). Prior to human settlement Santiago Island had a large population of land iguanas. Scientist, Charles Darwin described the lowlands of Santiago as saturated with "infinite numbers of a large herbivorous lizard" during his expedition in the 1835 (Darwin, 1845). 17th century whalers introduced several mammals including, cats, dogs, rats, pigs, goats and donkeys (Jackson, 1993). The introduction of these mammals is considered the driving force behind the extinction of land iguanas on Santiago. Another endemic species the rice rat (*Nesoryzomys swarthy*) on Santiago was also considered extinct since 1906, however, the species was rediscovered in 1997 (Cruz et al., 2005). In 2005, introduced goats, donkeys, house mice and the smooth-billed ani (*Crotophaga ani*) were still present on Santiago (Cruz et al., 2005). According to Cruz et al. (2005) all pigs were eradicated from the island in 2000, this is an estimated 18,800 pigs. Soon after, all donkeys and feral goats were also completely eradicated as part of "Project Isabela," which launched in 1997. A total of 79,579 goats were killed between 2001 and 2006. Santiago Island has been free of mammalian predators and mammalian grazers since 2006, with the exception of *Rattus rattus*. The vegetation on Santiago island has shown a noticeable improvement with the absence of mammalian grazers (Carrion et al., 2011). However, the restoration of Santiago Island is not considered complete. With the removal of introduced mammals (except *R. rattus*) it is possible to reintroduce locally extinct species such as the land iguana. Land iguanas act as ecosystem engineers, facilitating seed dispersal and thus helping to revegetate the land (Traveset et al., 2016). Reintroducing land iguanas onto Santiago will help to restore the island, extend the geographic range of the species and boost the global population size of the species thus improving its conservation status.

THE PRESENT STUDY

In 2014 the IUCN SSC Iguana Specialist Group met in Puerto Ayora, Galapagos Islands to identify knowledge gaps for Marine, Santa Fe (*Conolophus pallidus*), Pink and land iguanas in the Galapagos Islands (iucn-isg.org). Among the priorities identified was the need to assess the current population sizes and threats of land iguana populations and potential source populations for a reintroduction of the land iguana to Santiago Island (Karl Campbell, Island Conservation Pers. comm.).

In 2016, I was invited to participate in this project to: a) compare morphometrics of land iguanas on islands where feral cats occur and cat-free islands; b) estimate the current land iguana population sizes on Fernandina and North Seymoor; c) review the IUCN conservation status of the land iguana with the most up-to-date information and d) identify priorities for future research.

These overall aims were met with a combination of field research and analysis of unpublished data provided by biologist Alizon Llerena whom conducted extensive monitoring of land iguanas from 1998 to 2012, and that form part of the most recent assessment of iguana populations in the Galapagos Islands (Marquez et al., 2010). These data was collected while A. Llerena worked at the Charles Darwin Research Station in Puerto Ayora, Santa Cruz and in conjunction with the Directorate of the Galapagos National Park. A. Llerena also provided extensive background information on handling techniques for land iguanas, local threats to the populations and personal observations of behaviour.

This thesis addresses key gaps in knowledge identified by the Directorate of the Galapagos National Park that will assist in the development of a strategy to translocate land iguanas to Santiago Island in 2019. At the onset of this thesis in 2016, the date for the translocation of land iguanas was expected to be considerably later. However, as information about land iguana populations resulting from this thesis became available, and resources and a strategy to eradicate black rats (*Rattus rattus*) from North Seymour also became available, a bold plan to move 2000 land iguanas accelerated between May 2018 to December 2018. With the first translocation occurring in January 2019. With regards to land iguanas the key gaps in information were: which land iguana population can act as a source for a large number of individuals?, would land iguana populations experiencing cat predation of juveniles be suitable as sources?, what is the most reliable method to survey land iguanas? and what is the approximate carrying capacity of Santiago Island for land iguanas?

In addition to these, it became apparent that the IUCN conservation of the species required revision and update and that a habitat assessment on Santiago Island was necessary prior to planning a translocation of the species.

Study species

The Galapagos Land Iguana (*Conolophus subcristatus*) is an endemic, island-dwelling species. Although the physiology and genetics of land iguanas have been researched extensively (Costantini., 2005, 2009; Ellis & James., 1978; Franco et al., 2011; Gentile, Ciambotta & Tapia., 2013; Hofkin at al., 2003) there is limited information on their conservation. The few articles that focus on conservation of land iguanas are centred on the eradication of invasive mammals (Fabiani et al., 2011; Harper, Zabala, & Carrion, 2011; Phillips et al., 2005). The only published article evaluating the local population sizes and threats of land iguanas is by Marquez et al. (2010). Land iguanas are one of four species of Iguanids belonging to the Galapagos Islands, Ecuador. Its conservation status according to the IUCN is "Vulnerable". However, this classification has not been reviewed since 1996 and currently there is no formal evaluation of their population size or demographic trends across their distribution. The IUCN assessment of land iguanas has near to no information and of the information that is available is outdated. The IUCN Red List is claimed to hold the most comprehensive information about the conservation status of all species ("The IUCN Red List of Threatened Species," 2017). It is a renowned tool for conservation decisions and management, thus the importance of an up-to-date and comprehensive assessment is crucial for any conservation action to take place successfully. Although many species across various taxa have up-to-date, comprehensive information, the land iguana does not. The result can have a hindrance on conservation planning and funding. All of the three iguana species belonging in the *Conolophu*s genus are protected under the Appendix II of "Convention on International Trade in Endangered Species of Wild Fauna and Flora" (CITES) (Gentile, Ciambotta, & Tapia, 2013). Although protected, there have been no conservation measures in place for land iguanas in over a decade, despite having 55% of the population threatened by feral cats (Cayot, 2008; Marquez et al., 2010). As a consequence, the extinction risk of land iguanas is heightened, which without substantiated and up-to-date information to prompt a conservation intervention could result in the loss of this species. Basic demographic data will aid conservation managers to understand the status of the species and to help establish a comprehensive management plan for the Galapagos land iguana.

The land iguana (*Conolophus subcristatus*) is amongst the oldest vertebrate lineages in the Galapagos. Iguanas in the Galapagos include two genera; *Conolophus* and *Amblyrynchus*, these are sister taxa that diverged from a common ancestor approximately 4.5 mya (MacLeod, Irisarri, et al.,

2016). The Galapagos iguanas are closely related to genera *Iguana* and *Cyclura* (MacLeod, Irisarri, et al., 2016; Pyron, Burbrink, & Wiens, 2013). *C. subcristatus* have the widest distribution of all terrestrial iguanas in the Galapagos. They occur on six islands in the Galapagos Archipelago, including; Fernandina, Isabela, Santa Cruz, North Seymour, Baltra and South Plaza (Fabiani et al., 2011). Land iguanas are diurnal creatures, found in the arid regions of the islands (Snell & Tracy, 1985). They are primarily herbivores and generalists, the *Opuntia* cactus is one of the more popular food choices for the iguanas (Traveset et al., 2016). They are also opportunistic feeders occasionally feeding on insects and carrion (Harper, Zabala, & Carrion, 2011). Land iguanas are fairly large lizards, reaching lengths of over a meter long, males have been found to weigh up to 13 kilograms (Marquez et al., 2010). Land iguanas exhibit sexual dimorphism from their late sub-adult to adult years. The sex of a juvenile is difficult to determine without the use of a cloacal probe. Males are characterized generally by larger body size, pronounced head scales and larger crests. The yellow colouration is also more pronounced in males (Switak, 1998). Natural predators of land iguanas include Galapagos racer snakes (*Pseudalsophis spp.*) and Galapagos hawks (*Buteo galapagoensis*), both prey on juvenile iguanas that are much smaller in size (Marquez et al., 2010; Werner, 1983). Introduced species such as goats, rats, dogs, pigs and cats threaten land iguana populations due to competition, habitat destruction and predation (Tzika et al., 2008).

Study sites

Study sites of my data collection include Fernandina Island and North Seymour Island in the Galapagos Archipelago. The Galapagos is part of the Republic of Ecuador, it is a set of volcanic islands in the Pacific Ocean that are situated approximately 10,000 km from the west coast of Ecuador, South America (Claudino-Sales, 2019). The climate in the Galapagos, although equatorial is seasonal and moderate, there is a warm, wet season from January to June and a dry, cooler season from July to December (Mueller-Dombois & Fosberg, 1998). Other study sites that were not visited as part of my study, however were included in my analysis were; Isabela Island, Baltra Island, Santa Cruz Island and South Plaza Island. I have used unpublished data provided by Alizon Llerena (previously at the Charles Darwin Research Station) which contains information on land iguanas from the islands named

previously. Land iguanas are found in the arid and low scrub regions of these islands. More detailed site descriptions will be given in the following chapter.

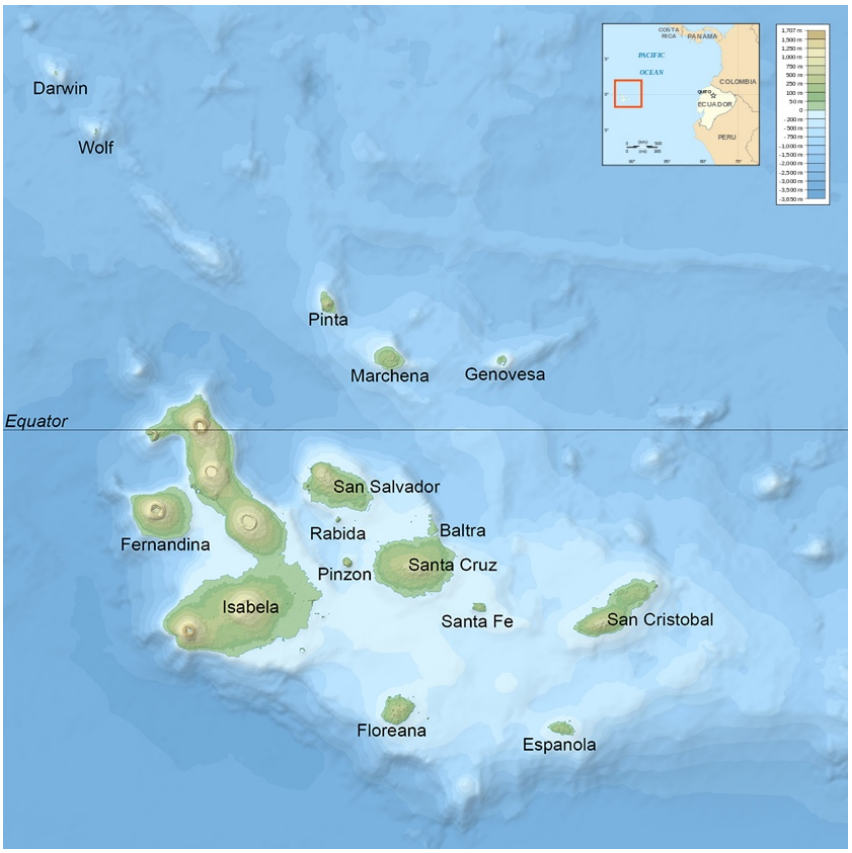


Figure 1.4: A Map of the Galapagos Islands, Ecuador. Modified from \ E. Gaba distributed under CC-BY 2.0 License

Methods

The methods for iguana handling in Chapters 3 and 4 are described below with images included. All field work was approved by the Galapagos National Park for my period of study, the necessary permit is included in Appendix B.

Upon capture, morphometric measurements were taken for each iguana. These include; Snout Vent Length (SVL), this is a measurement from the tip of the chin to the cloaca of the animal, see Figure 1.5 A; tail length, measurement from base of the tail to the tip of the tail and weight, a Pesola scale was used to weigh the iguana, with a rope noose around its chest that was dropped, see Figure 1.5 C. Body condition was calculated with Laurie & Brown's (1990) formula, see equation in Chapter 2 page 45. Additional notes on the condition of the iguana were also recorded. Lastly a Passive Integrated

Transponder (PIT) tag was inserted in the hind thigh of the iguana, see Figure 1.5 B, and a photograph was taken prior to release, see Figure 1.5 D.



Figure 1.5: Photographs showing iguana processing methods. Figure A shows the author measuring the Snout Vent Length (SVL) of an iguana held by a Galapagos National Park (GNP) Ranger. B shows author inserting a Passive Integrated Transponder (PIT) tag into the posterior leg of an iguana. C shows method used to weigh the iguana with a Pesola scale. D shows author taking a photograph of a processed iguana. E shows GNP ranger, Marcelo Gavilanes and author Kirtana Kumar holding land iguanas on North Seymour Island. Photos: L. Ortiz-Catedral.

THESIS AIMS

This thesis addresses key gaps in knowledge about populations of land iguanas in the Galapagos Islands to assist with the planning of a translocation of land iguanas to Santiago Island, explore the effect (if any) of feral cats on the various land iguana populations and provide recommendations for future research and management.

Objectives:

I. Provide an updated estimate of population status and characteristics of land iguanas in the Galapagos islands

II. Provide a population estimate of land iguanas on a remnant and translocated population on Fernandina and North Seymour Island

III. Review the IUCN threat category for land iguana

IV. Provide an overview of future research needs to galvanise the translocation of land iguanas to Santiago Island as well as provide research priorities for general land iguana conservation

CHAPTER 2

Population status of the Galapagos land iguanas (*Conolophus subcristatus*)



Figure 2.1: A Galapagos land iguana (*Conolophus subcristatus*) amongst succulents on North Seymour Island. Photo: K. Kumar

ABSTRACT

In order to accurately reflect the conservation status of a species, regular updates on demographic trends and population size estimates are required. The Galapagos land iguana has been the subject of episodic management since the 1970s in the Galapagos Islands and at present is classified as Vulnerable according to the IUCN. 55% of the global population is faced with the threat of predation of eggs and juveniles by feral cats. There is some uncertainty about the intensity of effects of introduced mammals of various Galapagos land iguana populations and is a growing cause of concern to conservationists and managers. Therefore, updated information about the local threats to iguana populations represent a priority for species management and to review international and national threat categorizations for this species. In this chapter, I compare a suite of demographic parameters between iguana populations occurring on islands where feral cats are present and islands with cats absent. I analysed unpublished data collected between 2004 and 2013 by staff from the Directorate of the Galapagos National Park and the Charles Darwin Foundation. My analyses includes the following islands: Fernandina, Isabela, Santa Cruz, Baltra and South Plaza. A morphological summary of each population is given along with a summary of the age structure and sex ratio. I revealed a significantly lower proportion of juveniles (1%) on islands were feral cats occur, than on islands free of feral cats (12%). Also, I discovered a significant difference in measures of body condition of adult iguanas, with individuals from cat-free islands showing lower average values. A potential explanation for this is the occurrence of a more diverse range of age and size classes among adults favoured by the absence of feral cats. I discuss urgent research needs to address the threat of feral cats to the viability of the Galapagos land iguana population. I conclude that while sourcing land iguanas for translocation to Santiago Island from cat-present populations could act as a "rescue" action, at present there is insufficient information to model the consequences of removing remnant mature individuals only. One possibility is significant decrease in local density (due to lack of recruitment of juveniles) and greater local extinction risk. Therefore, sourcing individuals from cat-free populations, with a better representation of all age and size classes appears as a more viable option.

INTRODUCTION

Species extinctions, and the associated loss of their evolutionary uniqueness, ecosystem function and even services to human societies, constitute one of the most alarming elements of the global biodiversity crisis (Barnosky et al., 2011; Case, Bolger, & Richman, 1992; Ceballos et al., 2015; Ceballos, García, & Ehrlich, 2010; Costello, May, & Stork, 2013; De Vos et al., 2015; Dirzo & Raven, 2003; Harnik et al., 2012; Hughes, Daily, & Ehrlich, 1997; Pimm et al., 1995, 2014; Smith et al., 1993). Numerous studies have measured the rate of species extinctions and the number of species becoming extinct, as well as their chronology (Barnosky et al., 2011; Case, Bolger, & Richman, 1992; Ceballos et al., 2015; Ceballos, García, & Ehrlich, 2010; McKinney, 1999; S. L. Pimm et al., 2014; Smith et al., 1993). Ultimately, the extinction of species represents the death of the last known individual. But until recently, the extinction of populations, which precedes the extinction of a species, has received less attention.

The loss of populations conveys various detrimental effects on ecosystem services and functioning. For example, declines in frugivore biomass worldwide represents a loss in the valuable ecosystem service of seed dispersal provided by 'healthy' populations of these species (Valiente-Banuet et al., 2015). Further, these losses can alter plant communities in a way that some plant taxa experience an ecological release, whereby there is an explosion in their numbers in response to decreased frugivore pressure (Ceballos, Ehrlich, & Dirzo, 2017). Thus, conservationists and managers have a duty to focus and document the processes that lead to local population extinctions as well as the loss of species. Understanding the status and pressures of various populations of a given species, can provide valuable information on the local extent of threatening processes, which in turn can be factored in, when assessing the threat status of a given species, following IUCN criteria.

For example, the African leopard species (*Panthera pardus pardus*) is widely distributed occurring in several countries within Africa as well as occurring in South Africa. Populations are patchy with limited dispersal opportunities. Not all population patches are protected (Jacobson et al., 2016). For instance there is less human-wildlife conflict with leopards in protected game reserves within South Africa, consequently the extinction risk of African leopards in protected game reserves is lessened due to the decreased human-mediated mortality. On the contrary populations that are outside the game reserve or may be on the edge of the protected area have increased extinction risk, as they are prone to

legal and illegal hunting, thus have higher mortality rates (Balme, Slotow & Hunter., 2009). Additionally, research on African leopards is biased towards populations that occur in protected areas further increasing the risk of extinction of leopards that occur outside protected areas (Dickman & Marker., 2005). Furthermore Leopards of the same species occur in low densities in Namibia, as a result of persecution from farmers due to livestock predation from leopards areas (Dickman & Marker., 2005). Lastly, another article reported greatest range loss of African leopards in North and Western African regional populations (Jacobson et al., 2016). A growing conservation issue is the lack of connectivity between sub-populations of a species. Sub-populations of a species' metapopulation can exhibit varying levels of extinction risk due to a number of factors both external and internal, such as anthropogenic habitat fragmentation and small population size respectively (Akçakaya & Ginzburg, 1991). For example Isaac & Cowlishaw (2004) reviews the threats faced by several ape species and found that there is a variation in vulnerability to threat type within the same species such as agriculture posing a higher degree of threat when compared to forestry, different local populations are faced with either one or the other resulting in either a high or lower risk of extinction when compared to neighbouring populations.

Prior to managing wild populations and intervening with conservation measures, an understanding on the status and state of the individuals in single population must be achieved in order to manage each population accordingly as well to assign the appropriate conservation measures tailored to the populations' needs. Additionally, understanding the current state of sub-populations can assist with the prioritization of conservation efforts by identifying which populations are subject to a higher threat of extinction and are in need of more urgent conservation management. Thus demographic studies on populations can provide some of the first insights into the specifics of a population.

A disproportionally large number of critically endangered species occur on islands (estimated at 37%) (Spatz et al., 2017). Additionally, islands hold a large proportion of endemic species, which include some of the rarest (sometimes single populations) and most threatened species in the world (Kier et al., 2009). The vulnerability of island populations and species to extinctions is greater than mainland species. Restoration, habitat protection, removal of introduced fauna and translocations are some common and powerful methods used to manage populations on islands (Akçakaya, Mills, & Doncaster, 2007). For species that occur on multiple islands and face different degrees of threat, it is

important to document as best as possible their population status in order to prioritize conservation action. The Galapagos land iguana is an ideal example of this scenario. The species occurs on six different islands and there is evidence of diverse phenotypes adapted to the local conditions (Snell, 1984). Further, there is genetic evidence supporting distinct conservation units (Tzika et al., 2008). Their dispersal abilities are limited between the islands and thus have never been recorded to have exchange of individuals between island populations. However, to some extent, since the 1970's land iguanas have been managed as metapopulation. For instance, since the late 1980 the population of land iguanas from Cerro Dragon have been supplemented with individuals hatched on Venecia (Fabiani et al., 2011). In line with metapopulation theory, this action can be seen as a key to increase the size or persistence of the individual or local populations, which results in benefiting the population as a whole. (Resit Akçakaya, Mills, & Doncaster, 2007).

Historical studies on land iguana populations have centred on genetics and phylogeny (Hofkin et al., 2003; Snell, Snell, & Tracy, 1984; Tzika et al., 2008). Genetic studies on wild populations play an understandably important role in determining populations that are in need of conservation intervention (Booy et al., 2000; Derek et al., 2004; Malone, Knapp, & Taylor, 2003). However, genetic studies have often overshadowed literature on demographic studies of land iguanas. It is widely acknowledged now that introduced feral cats have detrimental effects on iguana populations across their range (Iverson, 1978; Mitchell et al., 2002; Nogales et al., 2013; Palmeirim, 2007). With that being said, studies on the impact that feral cats have on populations of land iguanas are lacking, despite reports of previous population declines of land iguanas in areas with cats (Phillips et al., 2005; Tzika et al., 2008). Thus studies on the impact of feral cats will prove to be useful information for management of land iguana populations.

Similarly, demographic studies on populations are arguably another fundamental requirement in informing management decisions. Freake & DePerno (2017) present a study on managing Hellbender salamander (*Cryptobranchus alleganiensis*) populations and caution the necessity for demographic studies in conservation planning and have shown that a lack of a demographic data can lead to unfavourable outcomes. Having a multifaceted approach to conservation increases the likelihood of success in conservation operation. Demographic factors must be considered and incorporated alongside detailed baseline data as well as current and previous knowledge when developing future conservation

plans for the Galapagos land iguana populations. Conservation science is multifaceted and requires the integration of a wide range of elements including genetics, evolutionary history, demographics, political interests and socioeconomic circumstances (Hall & Fleishman, 2010). If certain key aspects are neglected in managing wild populations, one can expect an unsuccessful outcome. Further, decisions informed from only a single facet have shown to have negative repercussions on a population (Hedrick et al., 1996; Lande, 1988; Szucs et al., 2014; Zenboudji et al., 2016). The land iguana exists in at least 11 known populations across the Galapagos archipelago. The last population study of land iguanas was conducted by Marquez et al. (2010). The study covered eight populations of land iguanas and was an informative study just giving a summary of the features of each population. Conservation status, threats and future projections for the populations were not investigated or discussed.

Although the total population size of land iguanas is unknown, it is estimated at over 10,000 mature individuals, and have been classified as 'Vulnerable' by the IUCN Red List (Marquez et al., 2010). However, this number is not evenly distributed across known populations and masks the higher risk of iguana populations in human inhabited habitats. This classification suggests that major losses of land iguana populations may not occur for some time. However, the extinction or a major population collapse, as has been observed in other iguana species which occupy island habitats (refer to Chapter 1), an intervention is required to expand the size of the current populations and safeguard them from such events (Myers et al., 2000). Due to their relatively high numbers, land iguana are also an ideal candidate for creating new populations, without risking the loss of source populations (Alberts, Carter, & Hayes, 2004; IUCN/SSC, 2013).

The role of land iguanas as seed dispersers make them essential species in the ecosystems that they inhabit (Traveset, 1990; Traveset et, 2016). Land iguanas in the Galapagos Islands are considered to provide key ecosystem services on the islands they occupy and are known to have contributed to the revegetation of Fernandina following eruptions (Hendrix, 1981). Their scats contain numerous native plant seeds which are spread when iguanas move around the habitat. On Fernandina, land iguanas travel long distances for an iguanid species which likely aids further dispersal of native plants (Hendrix, 1981). Both Fernandina and Santiago are large islands and are similar in size with an area of 64,200 hectares and 58,500 hectares respectively (Cruz et al, 2009; Werner, 1983). Since the introduction of primarily grazing mammals to Santiago Island, the vegetation has drastically reduced and is degraded (Cruz et

al., 2009). Further, the loss of land iguanas from Santiago Island meant losing one of the important seed dispersers of the island, thus the vegetative restoration process after the removal of introduced grazing mammals is slowed down without the aid of iguanas and their seed dispersal abilities. This suggests that if land iguanas were introduced to Santiago, see Chapter 1 and Chapter 5, they could potentially disperse across much of the island and assist revegetation by spreading native plant seeds in their scat.

This chapter presents the demographic information of land iguanas from six different populations on five islands, by integrating data from 2004 until 2013. My aim is to update the population status of land iguanas from each population as well as investigate whether cats have a significant effect on land iguana populations. This will address the current knowledge gaps in land iguana research. By comparing the population status of these populations and examining whether the presence of feral cats have significant effects on land iguana populations, this chapter can assist the Galapagos National Park (GNP) in identifying applicable management interventions suitable for potential future translocations or other modes of management for land iguanas.

Although I could not explore the effects of feral cats on land iguana populations experimentally, I present an analysis of the effects of feral cat presence on the proportion of juveniles as a proxy of the effects of these introduced predators on a vulnerable age class. One of the options for sourcing land iguanas for translocation to Santiago Island is the "rescue" of adult iguanas from Isabela and Santa Cruz Islands (an upscaling of the management currently in place for the Cerro Dragon-Venecia populations). However, there is uncertainty about the representation of land iguana age classes on cat-present and cat-free islands. This Chapter therefore focusses on potential effects of the presence of cats on estimates of body condition for land iguana and proportions of age classes using data collected during an 9-year period.

Objectives

i.) Provide an updated demographic status of land iguana populations in the Galapagos archipelago integrating data from 2004 till 2013, with six populations, from five different islands

ii.) Examine indirectly the effects of feral cats on land iguana populations by comparing body condition index, size classes and population structure indexes for cat-absent and cat-present populations

METHODS

Study sites:

Baltra Island (0°30′S 90°01′W)

Area size: 2100 ha. Baltra is home to the Galapagos airport and military airbase, situated near the northern coast of Santa Cruz Island. It is a low and flat island, with minimal and sparse vegetation. Low growing shrubs and Prickly pear cacti (*Opuntia*) are the dominant vegetation type (Hamann, 1979; Phillips et al., 2005).

Fernandina Island (0°22′S 91°33′W)

Area size: 64,200 ha. Fernandina Island is the youngest island, emerging approximately 70,000 years ago (Schatz, 1998). It is also the third largest island and western most island of the Galapagos archipelago. It consists of an active shield volcano that last erupted in 2018 (Vasconez et al., 2018). It is one of the few undisturbed islands with no introduced mammals. Fernandina is heterogenous with varying levels of elevation and vegetation (Werner, 1983). The western flank of the island as well as the land surrounding the caldera rim has extensive vegetation, with the most common form of vegetation cover being trees, lianas and grass shrubs, this is the humid zone. Approximately 80% of the island consists of barren lava fields with extremely sparse and limited vegetation, mostly in the lower parts of the island (Adsersen, 1989; Heleno et al., 2011; Hendrix, 1981). Old lava flow has formed crevices on

the slopes of this island, where vegetation appears to grow opportunistically These patches of growth as called "*kipukas*" (Mueller-Dombois & Fosberg, 1998).

Isabela Island (00°30′S 91°04′W)

Area size: 467,000 ha. Isabela is the largest island of the archipelago, it consists of six volcanoes, of which five are active. Vegetation patterns here are heterogenous and similar to Fernandina with several lava fields. The vegetation is relatively low in terms of tree height and is concentrated in areas called "humid-zones" (Adsersen, 1989). These humid-zones like in Fernandina are fairly sparse and situated around the crater rims. Isabela contains several different introduced mammals and plants which threaten native fauna and flora (Eckhardt, 1972).

Santa Cruz Island (Cerro Dragon) (0°31'32.8"S 90°29'18.1"W)

Area size: 15 ha. Cerro Dragon is situated in the north-western side of Santa Cruz island. It consists of a trail that is approximately 1.6 km long. It is a popular site for tourists as it holds a small population of land iguanas. Feral cats are present here (Fabiani et al., 2011).

Santa Cruz Island (Venecia) (0°51′S 90°38′W)

Area size: 14 ha. Venecia is a small islet of Santa Cruz Island, it is situated two kilometres north of Cerro Dragon beach (Snell, 1994) and holds an introduced population of lands iguanas (Cayot, 2008).

South Plaza Island (0°58′S 90°16′W)

Area size: 13 ha. South Plaza Island is situated east of Santa Cruz Island. It is oldest island in the Galapagos archipelago at approximately 4.2 million years old (Schatz, 1998). Although the island is small it has an extremely dense land iguana population. This is the only island where breeding occurs between land iguanas and marine iguanas, producing a hybrid iguana (Rassmann, Trillmich, & Tautz, 1997). The likely cause is that land iguanas are found closer to the shoreline on South Plaza island than any other island. The island is sparsely covered by *Opuntia* cacti as well as succulents. The western part of the island contains low growing thorny scrubs (Carpenter, 1969).

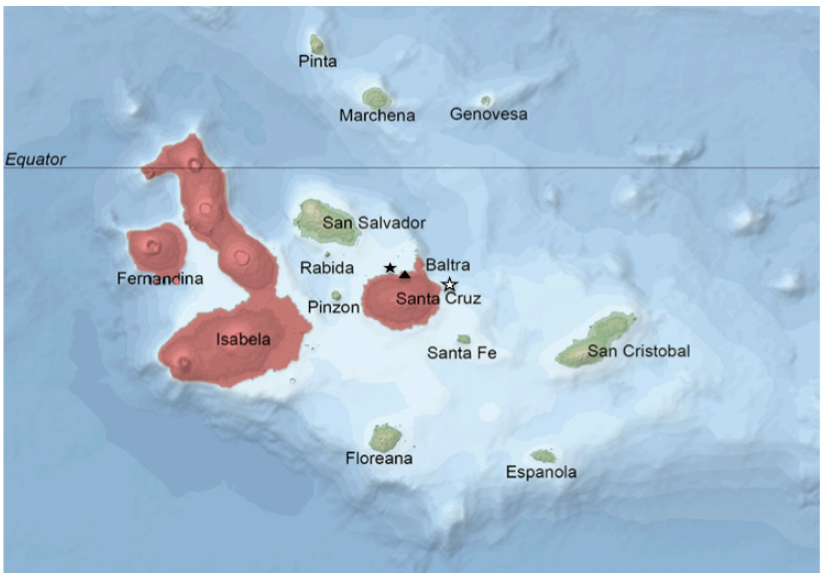


Figure 2.2: Study sites on the Galapagos Islands (Northern Islands Wolf and Darwin are excluded). The study islands (see text) are shown in red. Black star: Islet Venecia; white star: South Plaza; black triangle: Cerro Dragon. Modified from \ E. Gaba distributed under CC-BY 2.0 License

Data collection

Data were collected from 2004 to 2013 by Alizon Llerena, then a Herpetologist at the Charles Darwin Research Station and staff from the Galapagos National Park. The dataset represents six populations from five different islands that were monitored for one or more years (see Table 2.1). Collectively, the dataset consists of 1,271 iguana records. See Figure 1.5 on page 28 for iguana processing methods and description. Morphological measurements were obtained upon capture and body condition was calculated with Laurie & Browns (1990) formula which was used first on marine iguanas, refer to equation and text on page 44 of this Chapter.

Table 2.1: Years monitored for each land iguana population

	Fernandina	Isabela	Cerro Dragon	Baltra	South Plaza	Venecia
Years monitored	2004 2006 2012	2012 2013	2011	2007 2012	2012 2013	2011

Data analysis

There are morphological differences between populations as well as unique genetic groups identified within *Conolophus subcristatus.* Tzika et al. (2008) reported three genetic groups (meaning populations that share similar genetic makeup) in populations of land iguanas. These groups include;

Plaza Sur, Isabela + Fernandina and Santa Cruz. The Baltra and North Seymour population show mixed ancestry with Isabela + Fernandina and the Santa Cruz group, as both the Baltra and North Seymour population have been a subject to a number of translocations. Additionally Snell, Snell & Tracy (1984) reported morphological differences in populations. Their study included populations from all islands barring North Seymour, they reported that individuals from South Plaza island were the smallest individuals with notably smaller SVL sizes, and individuals from Baltra island represented the largest iguanas with large SVL values.

With these population differences in mind, I provide descriptive statistics on the data available for this chapter following this text. This is done prior to our analysis on the differences between cat-present and cat-absent populations to account for the already existent different morphological features of each island.

Since the sample sizes and temporal sampling between locations varied, I tested for normality of data sets using the Shapiro-Wilk test on R version 3.4. I also tested for homogeneity of variances using Levene's test. Data are non-normally distributed ($P < 0.0005$). I therefore used non-parametric tests to investigate differences between populations. I used a Kruskal-Wallis test and the Pairwise Wilcox test on R and contrasted three variables SVL (Snout Vent Length), mass, and body condition. Males and females were separated for this part of the analysis and only adult individuals were used. I excluded from analyses individuals with abnormally short tails, most likely lost to fights or injuries (n= 101).

A total of 1,112 iguanas were included in the analysis. With 523 males and 589 females. Table 2.2 shows a summary of land iguanas from each site, both male and juvenile percentage are given for each population. The Kruskal-wallis test results for comparing the SVL of each population showed highly significant differences between groups for both males and females (males (n=523) ($\chi^2 = 274.31$, $P < 0.0001$)), (females (n= 589) ($\chi^2 = 397.37$, $P < 0.0001$)). Iguanas from South Plaza Island were found to have the smallest average SVL (females (n= 213) 36.00 cm; males (n= 151) 40.78 cm). Populations with intermediate SVL size included; Baltra females (n= 49) 41.83 cm; Baltra males (n= 72) 47.45 cm, Fernandina females (n=230) 43.33 cm; Fernandina males (n=140) 49.50 cm and Cerro Dragon females (n=26) 46.91 cm; Cerro Dragon males (n=9) 51.25 cm. The largest individuals were found on Isabela Island (female (n= 71) 45.03 cm; male (n= 151) 51.26 cm).

Kruskal-wallis test results for both mass and body condition index also show a highly significant difference between groups for both sexes (male mass (n=523) ($\chi^2 = 299.75$, $P < 0.0001$)) and (female mass (n= 589) ($\chi^2 = 379.81$, $P < 0.0001$)); (male body condition (n=523) ($\chi^2 = 131.74$, $P < 0.0001$)) and (female body condition (n= 589) ($\chi^2 = 116.06$, $P < 0.0001$)). Iguanas with the smallest average mass were found on South Plaza Island (females (n= 213) 1.84 kg; males (n= 151) 2.87 kg), this corresponds to their small average SVL size. Iguana populations with intermediate average masses include; Fernandina females (n=230) 3.42 kg; Fernandina males (n=140) 5.35 kg, Baltra females (n= 49) 3.52 kg; Baltra males (n= 72) 5.29 kg and Cerro Dragon females (n= 26) 3.70 kg; Cerro Dragon males (n=9) 4.81 kg. Isabela Island had the largest average mass with females at (n= 71) 4.50 kg; males (n=151) 6.75 kg.

Populations of iguanas with the lowest average body condition include Cerro Dragon: females (n= 26) 35.94; males (n= 9) 35.60, South Plaza females (n=213) 39.52; males (n=151) 40.83. Those with intermediate body condition values include the Fernandina population: females (n= 230) 41.94; males (n= 140) 43.99. Populations with the highest body condition values include those from Baltra: females (n= 49) 47.76; males (n= 72) 48.42 and Isabela: females (n= 71) 49.82; males (n=151) 50.22. The average body condition calculated for every population indicates healthy individuals in terms of the body condition index criteria. According to Costantini et al. (2005) a body condition < 25 is considered as an individual which is unhealthy, in starvation or dying. All five populations have an average body condition over 30. A series of Box-plot graphs (Figures 2.3-2.5) following this text show the range exhibited for every population for each factor of interest outlined previously. The left side of the boxplot graph starts with the western most island (Fernandina) and ends with the eastern most island (South Plaza) on the right side of each graph. Sexual size dimorphism is apparent with males exhibiting longer and heavier bodies than females. Body condition is fairly similar for both sexes.

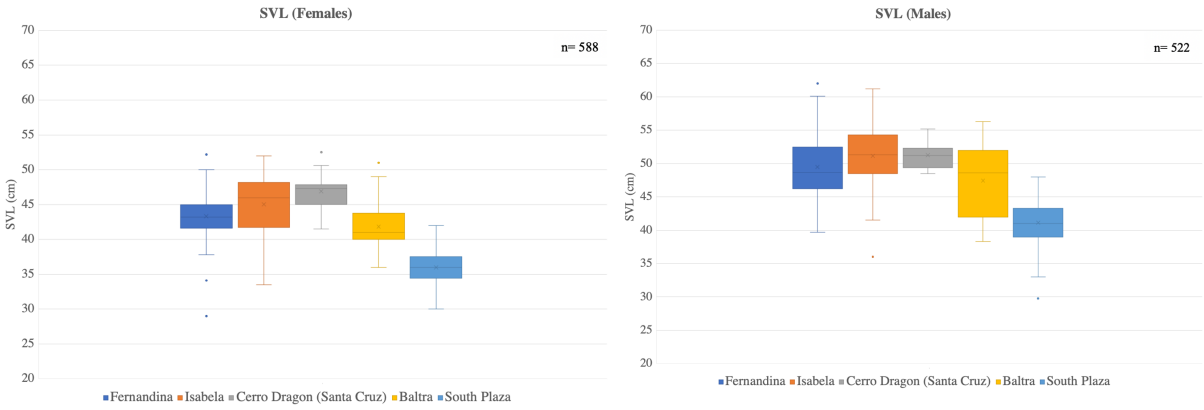


Figure 2.3: A comparison of Galapagos land iguana (*Conolophus subcristatus*) SVL (cm) measured at each study site.

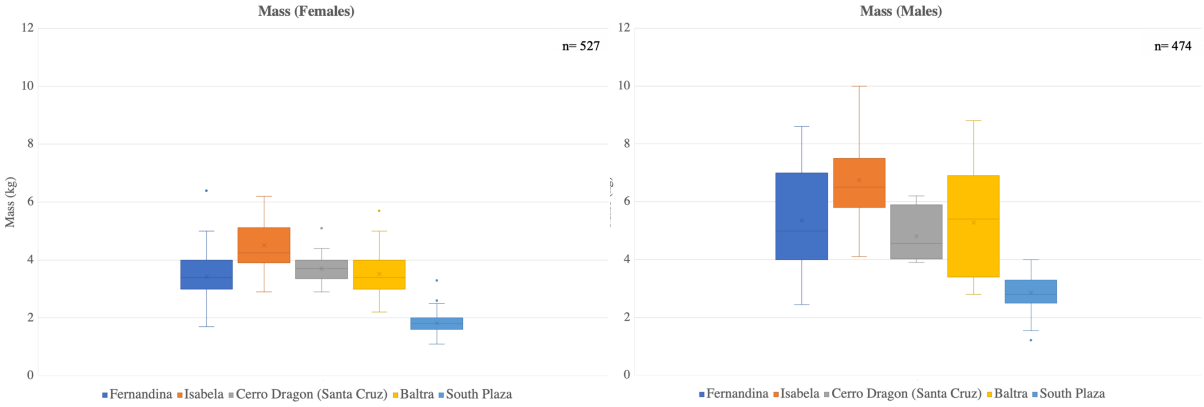


Figure 2.4: A comparison of Galapagos land iguana (*Conolophus subcristatus*) mass (kg) measured at each study site.

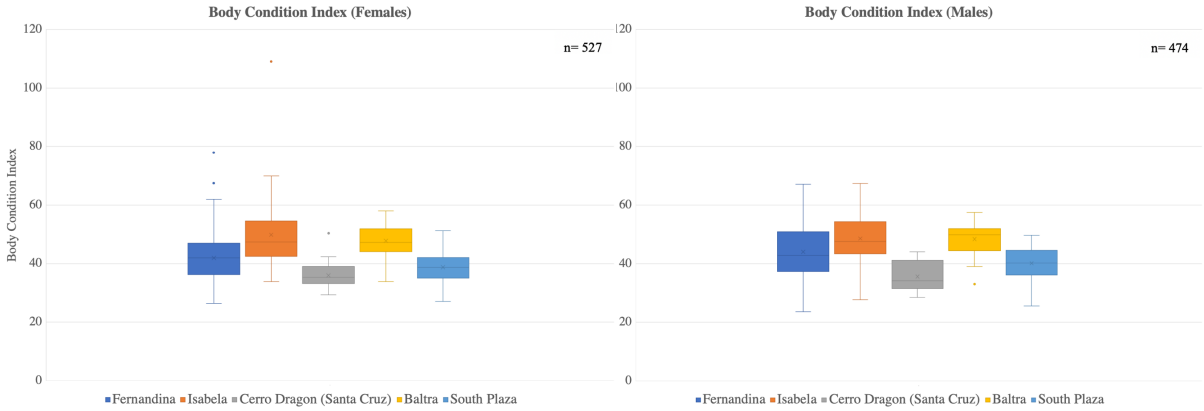


Figure 2.5: A comparison of Galapagos land iguana (*Conolophus subcristatus*) body condition indices given by Laurie & Brown's (1990) formula measured at each study site. The body condition index gives relative measure of 'fatness' of an individual. Indices under 25 are considered as starved and those over 60 are at maximum fatness.

Table 2.2: A summary of Galapagos land iguana (*Conolophus subcristatus*) morphometric measurements and population structure from six study sites in the Galapagos Islands.

	Fernandina	Isabela	Cerro Dragon (Santa Cruz)	Baltra	South Plaza	Venecia
n (males)	140	151	9	72	151	1
n (females)	230	71	26	49	213	6
Total	370	222	35	121	364	7
SVL (cm) (males)	49.50	51.26	51.25	47.45	40.78	47.50
SVL (cm) (females)	43.33	45.03	46.91	41.83	36.00	46.88
Mass (kg) (males)	5.35	6.75	4.81	5.29	2.87	4.40
Mass (kg) (females)	3.42	4.50	3.70	3.52	1.84	3.50
Body Condition Index (males)	43.99	50.22	35.60	48.42	40.83	41.06
Body Condition Index (females)	41.94	49.82	35.94	47.76	39.52	33.54
Male percentage (%)	39.05 %	68.02 %	26.47 %	59.50 %	42.74 %	14.29 %
Juvenile percentage(%)	3.65 %	0.89 %	2.94 %	15.38 %	5.37 %	19. 44%

n, sample size; **SVL**, Snout Vent Length (cm), **Body condition index**, relative measure of 'fatness' of an individual.
Note: Venecia is shown in red, as it is not to be considered as a true representation, due to the small sample size of 7 individuals, additionally most individuals from this sample were newly translocated juveniles.

I compared population structure and morphology of populations with cats present with populations with cats absent. at the time of data collection. To compare the morphological differences between the two groups of land iguana populations. I used a Welch two-sample t-tests for unequal variances (McDonald, 2009). Data were tested for normality with the Shapiro-Wilks test, and showed that the two categories are normally distributed ($P > 0.05$). A Chi-Squared test was used for comparing proportions.

For this part of the analysis I pooled males and females together, as the main factor of interest was size and not sex. Thus I have pooled males, females and all age classes including those with unknown sex in this part of the analysis.

Body condition index was calculated for all populations. Body condition index was calculated with Laurie & Brown (1990) index given for marine iguanas:

$$\text{Body Condition Index} = \frac{\text{Body mass}}{\text{SVL}^3}$$

This formula is used to calculate the "thinness" or 'fatness' of an individual which is used as a proxy for relative body condition (Laurie & Brown, 1990). This index has been used effectively on marine iguana studies in the past, as well as one land iguana study. Iguanas with indices < 25 are considered as starved and those at ≥ 60 are at maximum fatness (Costantini et al., 2005).

RESULTS

I have pooled data from all populations into two groups: cat-present and cat-absent. Populations with cats present include Isabela Island and Cerro Dragon on Santa Cruz Island. The remaining populations; Fernandina, Baltra, and South Plaza are free of cats. Iguanas from islands where cats occur have significantly larger SVL measurements than iguanas from cat-absent islands, regardless of sex (SVL cat-present (n = 238) 48.95 cm; SVL cat-absent (n = 878) 42.87 cm); Welch's two-sample t-test ($t = 16$, $P < 0.0001$). Iguanas from cat-present islands had significantly higher scores for body condition

estimates than iguanas from cat-free islands (Body condition cat-present (n = 238) 48.11; Body condition cat-absent (n = 878) 42.30) ; Welch's two-sample t-test (t = 7.5, P < 0.0001).

Populations on cat-present islands have significantly lower proportions of juveniles than cat-absent populations. On islands where cats are absent, the juvenile age/size classes on average represent 12% of individuals, while on cat-present islands this value is 1% on average ($\chi 2 = 21$, $df = 1$, P < 0.0001). Table 2.3 shows the proportions of juveniles per islands.

Table 2.3: A comparison of Galapagos Land Iguana (*Conolophus subcristatus*) populations measured at study sites with cat populations present versus study sites with the absence of cat populations

	Cats	No cats	Test statistic	P value
N	238	878	-	-
SVL (cm)	48.95	42.87	$t = 16$	< 0.0001
Body condition	48.11	42.30	$t = 7.5$	< 0.0001
Juvenile percentage (%)	1 (n= 258)	12 (n= 1,013)	$\chi^2 = 21$	< 0.0001

DISCUSSION

Results from the analysis show highly significant differences in size and population structure between cat-present and cat-absent populations with cat-present populations exhibiting significantly larger SVLs when compared to notably lower SVL values in cat-absent populations as shown in table 2.3. This particular analysis includes iguanas from all age classes. It is unknown exactly how large an individual land iguana must be to avoid being preyed on by cats, however previous documentation on predation of West Indian rock iguanas from feral cats show they can prey on individuals up to 23.50 SVL cm (Iverson, 1978). In my analysis, the iguana with the smallest SVL from a cat-present population was 32.40 cm. This indicates likely predation on iguanas with smaller SVLs such as juveniles as no iguanas under 30.00 cm were recorded. In contrast, the smallest iguana from a cat-absent population was 13.00 cm SVL of juvenile size and there were approximately 55 iguanas with an SVL ≤ 30.00 cm.

It is likely that smaller individuals in cat-present populations are preyed on by cats this has led to a higher proportion of large individuals in the population; in cat-absent populations there is a more even proportion of small juvenile and large adult individuals thus the smaller individuals would bring the average down, explain the lower average SVL in cat-absent populations. However, earlier analysis in this chapter has revealed populations from Isabela Island (cat-present population) constitute the largest land iguanas across all islands and South Plaza (cat-absent population) have the smallest iguanas. Therefore, this result should be kept in mind as it may affect the averages produced in the cat-present and cat-absent population groups. As for body condition I also found a significantly higher average body index value in cat-present populations, refer to table 2.3.

The body index formula (SVL^3/mass) is proportional measurement that is body mass relative to SVL, quantifying the "fatness" of an individual. Individuals with large or intermediate SVL lengths with lower masses will show lower body indices indicating a thinner individual on average and vice versa for those with higher masses. The indices have no relation to the age of the individual, thus is purely just a value to indicate the level of fatness of the iguana at the time. The fact that western island populations have larger individuals does not necessarily mean that their body indices will be also larger because they are larger, so perhaps comparing the body indices is a fairer indication of the state of iguanas between cat-present and cat-absent populations than just their size, since size is known to vary across islands possibly due to evolutionary and genetic reasons. The analysis of body index between the islands shows that cat-present populations have more individuals with higher body indices suggesting that cats' prey on individuals with lower indices leaving a higher proportion of individuals with high body indices. Similarly, cat-absent populations will have a range of body indices with the absence of selective predation, this is the same case for the difference in SVLs.

Land iguanas are long lived animals, known to live for over 60 years (Jackson, 1993). Cat-present populations contain iguanas that have managed to survive cat predation throughout their juvenile and sub-adult years and have outgrown the threshold for cat predation. These resulting individuals in the sample from cat-present populations are therefore older individuals that are in optimal condition. A factor that may contribute to their exceptionally large body condition, as shown with Isabela island in table 2.2, is the possibility that individuals in this population have decreased intraspecific competition and densities as a result of the absence of many smaller and younger land

iguanas that have been depredated by feral cats. Thus there are more resources available for the remaining population which consequently increases their overall body condition. Decrease in intraspecific competition as a result of predation has been observed in other species. For instance predation pressure from a number of bottom-feeding fish on a population of an amphipod crustacean species (*Corophium volutator*) has known to reduce intraspecific competition of juveniles, whereby the fish heavily predate on adult *C. volutator* as they are much larger and more desirable prey. Episodic predation occurs in the summer, thus adult density is lowered resulting in reduced intraspecific competition of juvenile amphipods, such that they are able to easily find burrows within their birthing place as opposed to have to emigrate out of their birthing range when adult densities are higher in the other seasons (Herbert, 1989).

Perhaps the best indicator of juvenile predation from feral cats is testing the proportion of juveniles to adults in a given population. Consequently, I propose the results for this part of the analysis is the more accurate and fairer indicator of juvenile predation from feral cats. I determined that cat-present populations harbour a significantly lower proportion of juveniles (1%) than cat-absent populations (12%). This analysis includes data from the wet and dry season for both groups, so breeding, laying and hatching seasons were all included. It is highly likely that juvenile predation from cats is the cause for this extremely low proportion of juveniles in these populations, as in the case of other island iguanas. Juvenile predation leads to a reduction in juvenile recruitment thus these populations show features of an aging population. There are numerous examples of juvenile predation on iguana populations from a number of iguana species. According to Medina et al. (2011) 14 difference species of iguana are threatened by feral cats. Cat predation on populations of iguanas is especially dominated by the *Cyclura* genus of West Indian iguanas as well as *Brachylophus spp.* from Fiji (Medina et al., 2011; Nogales et al., 2013). Iguana populations from the Caribbean, Fiji and Galapagos have been a subject to local extinctions and major declines because of the introduction of cats as mentioned in Chapter 1, this renders cat-predation as one of the most harmful threats known to iguana populations across their range.

This study provides the preliminary basis for monitoring land iguana population as well as uncovering the populations that are at most risk. The most important demographic finding is the age structure differences in cat-present and cat-absent populations of Galapagos land iguanas. Although we

did not measure the presence of cats directly, we are able to indirectly quantify the impact of feral cats on Galapagos land iguana populations with our analysis. Assessments for iguana populations exposed to cats in the Galapagos is lacking, possibly due to the wide geographic range of both species There is no knowledge on population trends or numbers. Based on our findings of age structure and previous literature on cat predation on iguana populations, I conclude that land iguana populations with cat's present (Isabela Island, Cerro Dragon: Santa Cruz Island) are under significant threat. The proportion of juveniles alone is enough to cause concern. In fact populations on Isabela Island and Cerro Dragon could be critically threatened. If left in the current state it is likely iguana populations from these populations will experience continuous decline threatening the entire population. Conservation priorities should be directed towards these populations, and should instigate further research into the effect's cats have on populations on Isabela Island. Isabela is the largest island and contains 5 out of 11 land iguana populations in the Galapagos (Marquez et al., 2010), thus holds a sizeable amount of the entire land iguana population.

If cat predation can be minimized with predator control efforts on local populations of land iguanas in cat-present populations, then we can expect to see an increase in juvenile recruitment, decreasing the threat status of these local populations. Although feral cat control has been under documented (Nogales et al., 2004), there are some examples of feral cat eradications that have enabled populations to recover in a number of native species. This is discussed further in Chapter 5.

Annual monitoring and direct measures of the presence of cats is encouraged and could uncover population trends and show if there is any significant decline that may be a cause for concern. Based on my analysis I recommend the implementation of cat control in core areas of land iguana populations. Additionally, our results can indicate those populations that are potential sources for future translocations without harming the integrity of the land iguana population. The analysis in this chapter provides preliminary information on the demographic characteristics of the metapopulation of land iguanas, which can be used to model and make quantitative predictions about their risk of extinction under different scenarios.

Based on my analysis of the proportion of juveniles and body condition of land iguanas across a representative section of its geographic range, I conclude that while adult land iguanas co-existing with feral cats exhibit higher body condition indexes, the low local proportion of juveniles makes them

unsuitable for translocation in large numbers to Santiago Island. These findings coupled with the low densities on cat-present islands (see Marquez et al., 2010) suggest that the removal of land iguanas from cat-present populations could increase the risk of local extinction. Although I do not have sufficient data to model these Population Viability scenarios, I recommend a cautionary approach when sourcing individuals from cat-present populations. The data in Chapter 3 indicates that a translocated cat-free population (North Seymour) offers a better option given the present population structure and evidence of high local recruitment.

CHAPTER 3

Population estimates of Land Iguanas (*Conolophus subcristatus*) on a remnant and translocated population: Fernandina and North Seymour Islands

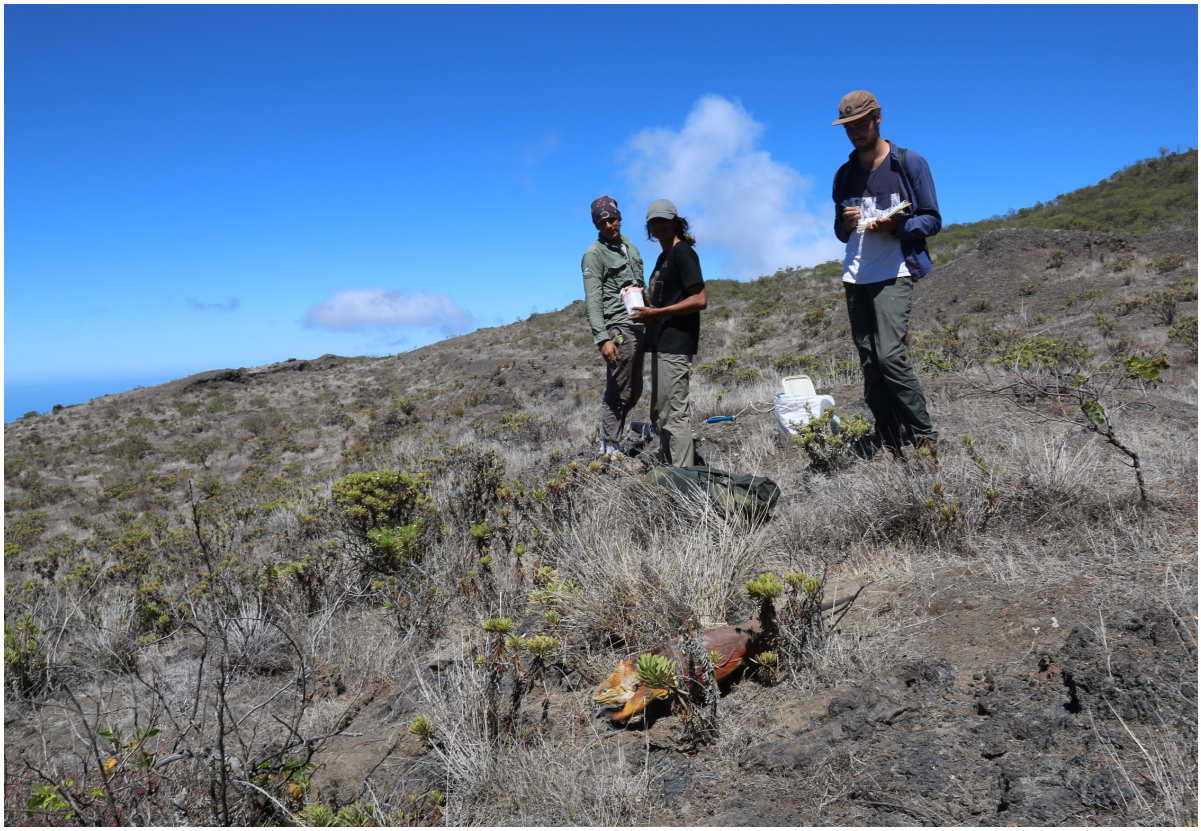


Figure 3.1: A land iguana (*Conolophus subcristatus*) is released by the field team after obtaining measurements and inserting a pit-tag for Mark-recapture estimates on Fernandina Island. From left: Johannes Ramirez (Galapagos Park Ranger); Kirtana Kumar and Harrison Sollis (volunteer). Photo: L. Ortiz-Catedral

ABSTRACT

Estimates of population size are fundamentally relied upon in conservation management, furthermore, population size plays a considerable role in determining the viability of a population. Prior to translocations, current population estimates and demographic variables of populations involved are necessary information for effective conservation management and successfully translocating the target species. This chapter presents population estimates for a remnant model population, Fernandina Island and a translocated population of North Seymour Island in the Galapagos islands using Mark-recapture and distance sampling The potential carrying capacity on Santiago Island, the destination for the translocation, is disclosed. I conclude that distance sampling is the more suitable method for surveying land iguana populations on small islands. Our estimates reveal a total of 7,979 iguanas on North Seymour Island with the distance sampling method and a total of 220,255 iguanas on Fernandina Island with the Mark-recapture method. Our comparison of parameters between the two population reveal a significantly lower body condition on North Seymour Island despite sharing a very similar SVL average to Fernandina Island, indicating a possibly over dense population, that can have consequences on the surrounding environment and wildlife. Finally I calculated the potential carrying capacity for land iguanas on Santiago Island at 838,302 iguanas. Our results are aimed at informing the Galapagos National Park on the translocation of land iguanas to Santiago Island.

INTRODUCTION

Conservation management actions implemented to reduce the extinction risk of populations and species requires a precise understanding of the population size and trends for the species of interest, as it can help determine the viability of a population (Reed et al., 2003), and determine priority actions. However, it is uncommon to account for every single individual in a threatened population, except in the case of extremely rare species. For instance, prior to its extinction in the field, a "population" of just three individuals of Po'ouli (*Melamprosops phaeosoma*) existed in the rainforests of Maui and every individual was accounted for in the intensive surveys for the species (Groombridge et al., 2004). The Po'ouli case is an example of extreme rarity, but more often conservationists need to estimate population sizes for species across larger geographic ranges or consisting of a larger number of individuals. In these cases statistical estimators that require the use of statistical models or indices of sub-samples of a population have been developed. They are inexpensive when compared to the effort of estimating absolute abundance and are currently common-place in conservation management (Pollock et al., 2002; Williams, Conroy, & Nichols, 2002). Estimators of population size (like distance sampling or Mark-recapture) take into account the realities of imperfect detection, sampling errors, and biological processes (i.e. seasonality) that might affect estimates of population size. Traditionally, these estimators operate within a set of assumptions developed to minimize the possibility of biased estimates (Grimm, Gruber, & Henle, 2014; Lettink, Armstrong, & Pryde, 2003; Ruiz de Infante Anton et al., 2013).

The challenge of choosing between available methods to survey populations is frequent in conservation management for various reasons. For instance, questions can arise on how many much funding is required to start the program or how many years or seasons can it be run as well as what constitutes a satisfactory margin of error. Thus, using more than one method can be useful to decide on which method is best suited for surveying a species of interest or to fine-tune survey effort against other management needs. For instance, surveying populations of the Giraffe (*Giraffa camelopardalis*) in Tanzania was traditionally restricted to aerial surveys with a small aircraft. More recently, more contemporary methods, such as Photographic capture-mark-recapture (PCMR) and distance sampling have been used in conjunction with aerial surveys to quantify the accuracy of the traditional method. This particular study concluded by suggesting the use of multiple survey methods to increase precision

and accuracy of population estimates (Lee & Bond, 2016). Similarly, a study conducted in New Zealand beginning in the 1990s used and compared multiple survey methods including aerial surveys, Mark-resighting, distance surveys and dung density sampling to estimate the size and growth rate of a feral horse (*Equus ferus caballus*) population in the Kaimanawa Ranges in the central North Island. The use of aerial surveys and Mark-resighting instead of Mark-recapture are typically used for larger animals. The study concluded that the Mark-resighting method was the most effective and gave the most accurate results, as well as providing demographic information for every individual (Linklater, 2001).

Determining population sizes of species of interest can also assist when deciding which of a suite of populations, represents the most appropriate source for individuals to establish or supplement another population via translocation. Translocations have been used commonly to assist the recovery of species and reduce their extinction risk. Overall, animal translocations have a high rate of failure. For instance Griffith et al. (1989) reported that only 44% of threatened native mammal and bird translocations for the purpose of conservation were successful between 1973 and 1986. This included translocations in Australia, Canada, Hawaii, New Zealand and the United States. Additionally it was found that herbivorous species tended to have higher success rates than carnivorous species translocations. Certain taxa are also sensitive to translocations which result in failure more often than not. Reptile and amphibian translocations have shown to have low success rates (19%) in comparison to other taxa. Some authors have even suggested that reptiles and amphibians are less suitable for translocations (Dodd & Seigel, 1991; Germano & Bishop, 2009).

Translocations or reintroductions have also been used for restoration purposes. Ecosystem functions have the ability to be restored following animal reintroductions. Nested within translocations, the process of rewilding can be found, this is a contemporary term for "species reintroduction to restore an ecosystem functioning" as described by (Seddon et al., 2014). A classic example of rewilding is described by many authors with the case of reintroducing Gray wolves (*Canis lupus*) to Yellowstone National Park in Northern America and its positive effects on the surrounding vegetation. After 70 years of absence, wolves were reintroduced to this region in 1995 (Ripple & Beschta, 2003). Following the reintroduction of these predators, the surrounding Elk (*Cervus canadensis*) population had decreased by 50% from 1995 to 2005. This also resulted in some behavioural changes in elk herds, where wolves were present, whereby elk had avoided certain areas due to the presence of wolves. The reduction in

the number of elk led to a widespread increase in vegetative growth of key plant species such as aspen (*Populus tremuloides*), willow (*Salix fragilis*) and cottonwood (*Populus spp.*), which were previously heavily grazed on by elk and was documented in a number of studies (Creel et al., 2005; Fortin et al., 2005; Ripple & Beschta, 2004; Ripple & Larsen, 2000). As Quaking aspen, is known to support a high number of species in comparison to conifer habitats also found in similar areas, the increase in the vegetative growth of these species had a positive effect on other species that inhabit and occupy the Yellowstone National Park area (Kimble et al., 2011). Restorative effects such as these can be achieved with animal translocations whether intended or just an indirect effect of the reintroduction. Proposals for the reintroduction of beavers (*Castor fiber*) to Scotland have been a subject of research for the last two decades, and are currently underway as beavers are known as ecosystem engineers (Gaywood, 2018; Stringer & Gaywood, 2016). Researchers suggest reintroducing species like these will have beneficial effects on the surrounding environment and neighbouring species. Despite the high failure rate of animal translocations (Bell & Herbert, 2017; Milligan et al., 2018). Several species of birds, mammals and reptiles have been successfully translocated . Depending on the species of interest and the circumstances of the translocation, some customary considerations prior to research include genetics, demography, habitat requirements and selection, disease, Population Viability Analysis (PVA), physiology and behaviour (Armstrong & Davidson, 2006; Buckland et al., 2014; Miller et al., 1999; Milligan et al., 2018).

An example of a translocation following a restoration rationale is the proposed reintroduction of Galapagos land iguanas to Santiago Island, in the Galapagos archipelago. Land iguanas occurred in Santiago Island up until the late 1800s, but extensive searches for these since 1906 have failed to locate living individuals and the species is considered extinct there (Werner, 1976; Marquez et al., 2010). Feral dogs, feral pigs and cattle have long been proposed as the culprits for the extinction of the land iguanas on Santiago Island (Steadman & Stafford, 1991), but the mechanism and chronology of their disappearance remains unknown (Steadman & Stafford, 1991). Early, “maverick” attempts at re-establishing land iguanas to Santiago Island in the 1970s involved the release of a few individuals from South Plaza (A. Llerena, pers. comm.). In the late 1970s, a single individual found on Santiago Island was removed and kept in captivity at the Charles Darwin Research Station (Werner, 1976). During a 30-year campaign to eradicate feral pigs from Santiago Island (Cruz et al., 2005) which involved

thousands of man-hours in the field, and covering practically the entire island, not a single land iguana was registered.

In 2014 at the IUCN SSC Iguana Specialist Group Meeting in Puerto Ayora, Galapagos Islands, there were discussions about research needs to plan a translocation of land iguanas to Santiago Island (K. Campbell, pers. comm.). Some of the perceived benefits of a translocation of that magnitude involved re-instating the role of seed dispersal and grazing played by land iguanas elsewhere in their range (Hendrix, 1981; Heleno et al., 2011; Traveset et al., 2016). One of the key populations identified for translocation was North Seymour, due to the apparent large population size there, however the lack of recent estimates there and uncertainty about the status of North Seymour land iguanas were commonly agreed research priorities (C. Sevilla and D. Rueda, pers. comm.).

The inter-island movement of land iguanas in Galapagos has a long tradition, spanning roughly 60 years (see Figure 1.3 on page 22 for reference of populations). It started in the 1930s when Perkins (1930) moved approximately 40 land iguanas from Baltra Island to North Seymour Island. Later, in 1982, 37 captive-bred land iguanas (parents sourced from Isabela Island) from the breeding centre at the Charles Darwin Research Station were released in Cartago Bay, Isabela (Reynolds, 1982). In 1987, 89 land iguanas (46 captive-bred; 43 wild-hatched) were translocated to Cerro Dragon in Santa Cruz Island (Snell, 1994). Further, in 1991-1992, 79 captive-bred land iguanas (parents sourced from North Seymour) were released on Baltra Island (Cayot & Menoscal, 1992) where the population was reduced to less than 50 individuals by the 1970s (Werner, 1976). Finally, since the 1990s, hundreds of land iguanas hatched on Venecia Islet have been translocated to Cerro Dragon (Snell, 1994), the most recent transfer involved a dozen of adult land iguanas in August 2017 (J. Ramirez, pers. comm.).

All the information from these past translocations has been useful to document aspects of the transfer of individuals and some information about their survival months after release, however the translocation to Santiago Island would most likely require thousands of individuals (K. Cambell, D. Rueda, pers. comm.) and therefore accurately estimating the population size of potential source populations, such as North Seymour is paramount.

In Chapter 2, I reported a significantly lower proportion of juveniles in cat-present populations of land iguanas and an associated higher body condition index for adult land iguanas, most likely the result of lower intra-specific competition for resources than on cat-free populations. Although

individuals of high body condition are desirable for translocation, the requirement of thousands of iguanas for translocation to Santiago Island renders cat-present populations unsuitable. This is because the removal of land iguanas from either Isabela or Santa Cruz, would effectively decrease the local population size of essentially non-reproducing populations. Further, a mixture of age classes is recommended for translocations and although there is no reliable system to age land iguanas, cat-absent populations appear to have a greater diversity of ages judging from the aspect of scales, size and tail ratios and overall colouration (A. Llerena, pers. comm.) (See also Appendix A).

An additional important piece of information highlighted during the 2014 IUCN SSC Iguana Specialist Group Meeting, is to see if there is a degree of similarity between population parameters and body condition estimates of land iguanas on North Seymour and a remnant reference population, such as Fernandina. In order to formally test this assumption I have conducted a number of tests to compare the population parameters of North Seymour and Fernandina Island detailed in this Chapter.

There have been few recent attempts at estimating population sizes of iguanas in the Galapagos Islands. Most studies have focused on Marine iguanas from 1990 to 2016 (Laurie, 1990; MacLeod, Unsworth, et al., 2016). For the land iguanas there have been two studies that have been published on the *Conolophus marthae* and *Conolophus subcristatus* populations of Volcan Wolf on Isabela Island, which occurred from 2009 to 2010 (Gabriele Gentile et al., 2016) and a separate study on the population size and density of all iguanas under the *Conolophus* genus (Marquez et al., 2010). However certain *Conolophus subcristatus* populations in Marquez et al's (2010) study were only sampled once on a single day with one method used, for instance their estimates for the iguana populations on North Seymour Island and Baltra Island were based on a single distance sampling survey on one day.

In this Chapter, I present an estimate of population size for North Seymour and Fernandina using Mark-recapture and Distance line transect surveys. This study is the first comprehensive population study on North Seymour Island after 27 years since the completion of Izurieta's study of iguanas on North Seymour island (Izurieta, 1991). Obtaining recent population estimates for the land iguana will be beneficial for long term monitoring of the species and informing the translocation. For land iguanas, our chosen survey methods were based on practicability and suitability of the method for the species: Mark-recapture and distance sampling. Both Mark-recapture and distance sampling are cost effective and feasible options with the time and budget constraints of this project. Each method

comes with its own limitations, which will be discussed later. Mark-recapture methods have been used on wide range of taxa including herpetofauna for estimating population size (Wanger et al., 2009). Based on previous studies, the combination of Mark-recapture and distance line transect surveys have proven to be the most suitable method for estimating population sizes of terrestrial lizard species (Burton & Rivera-Milan, 2014).

Objectives:

i.) Estimate the population size of land iguanas on one remnant and one translocated population: Fernandina and North Seymour respectively.

ii.) Provide an updated description of the population structure and morphological characteristics of both populations.

iii.) Estimate the available habitat for land iguanas on Santiago Island and model the carrying capacity of Santiago Island.*

**This objective was not fulfilled due to lack of sufficient funding to access sampling locations on Santiago Island in 2017 and 2018. Therefore, this objective is briefly explored in this Chapter and discussed as a possibility for further research in Chapter 5.*

METHODS

Study sites

Fernandina (0°22′S 91°33′W):

Area size: 64,200 ha. Fernandina Island is the youngest island, emerging approximately 70,000 years ago (Schatz, 1998). It is also the third largest island and western most island of the Galapagos Archipelago. It consists of an active shield volcano that last erupted in 2018 (Vasconez et al., 2018). It is one of the few undisturbed islands with no introduced mammals. Fernandina is heterogenous with varying levels of elevation and vegetation (Werner, 1983). The western flank of the island as well as the land surrounding the caldera rim has extensive vegetation, with the most common form of vegetation cover being trees, lianas and grass shrubs, this is the humid zone. Approximately 80% of the island consists of barren lava fields with extremely sparse and limited vegetation, mostly in the lower parts of the island (Adsersen, 1989; Heleno et al., 2011; Hendrix, 1981). Old lava flow has formed crevices on the slopes of this island, where vegetation has appears grow opportunistically. These patches of growth as called "*kipukas*" (Mueller-Dombois & Fosberg, 1998).

North Seymour (0°395592′S 90°287327′W):

Area size: 190 ha. North Seymour Island is north of Baltra and Santa Cruz Island. The island consists of a basaltic plateau with sparse vegetation, mainly dominated by *Opuntia echios* cactus and small trees of low growing shrubs. *Bursera malacophylla*, *Parkinsonia aculeate* and *Scalesia crockery* (Harper, Zabala & Carrion, 2011).The mid part of the island is rocky (personal observation). Introduced rats (*Rattus spp.*) are a common introduced species (Harper, Zabala & Carrion, 2011).



Figure 3.2: Photographs showing study sites Fernandina and North Seymour Island with land iguanas present. Top left shows North Seymour Island, Platforma. Top right photo shows land iguanas in their habitat on North Seymour. Bottom left shows Fernandina, with the crater in the back ground taken from near the shoreline. Bottom right photo shows land iguana habitat at the crater rim of Fernandina Island. Photos: K. Kumar

I estimated the population size of land iguanas on North Seymour and Fernandina Island using Mark-recapture and distance sampling. Using Mark-recapture surveys for estimating lizard populations has been one of the most common population estimation methods used for the last four decades. It has been used on Anolis lizard populations in the West Indies in the late 1970s when Mark-recapture/capture-recapture was a new method for estimating populations, since then it has been widely used on a number of lizard species globally (Heckel & Roughgarden, 1979; Tenan et al., 2013). For iguanas in particular, there have been long-term Mark-recapture studies on the Allen Cays rock iguanas in the Bahamas, which took place from the 1980s to 2000 (Alberts, Carter, & Hayes, 2004). There have also been Mark-recapture studies on the Anegada ground iguana, Mona island iguana, Fijian Crested iguana and more iguana species from other archipelagos (Mitchell, 1999; Morrison et al., 2009; Pérez-Buitrago, Sabat, & McMillan, 2010). Additionally, Mark-recapture allows researchers to gather morphological information useful to quantify age and size variation and body condition estimates.

Mark-recapture

Mark-recapture involves marking capturing individuals in a wild population, then releasing them back into that population, and subsequently recapturing some of the population which may or may not be marked. The concept centres on the number of marked individuals and their proportion in subsequent samples with the assumption that the subsequent samples represent the proportion of marked individuals in the population as a whole (Begon, 1979; Wilson, Hammond, & Thompson, 1999). The model I am using for estimating the population size is the Schnabel method with the Chapman modification (Chapman & Overton, 1966; Schnabel, 1938):

$$\widehat{N} = \frac{\sum_{t=1}^{S} (C_t M_t)}{\left(\sum_{t=1}^{S} R_t\right)+1} \qquad \text{(Equation 1)}$$

($\widehat{N}$) Estimated population size

(S) Number of sample events

(C_t) Number captured at t^{th} sample event

(M_t) Total number of marked individuals in the population during t^{th} sample event

(R_t) Number of recaptures during t^{th} sample event

The Schnabel method (Equation 1) had first originated from estimating fish populations and is now applied to a number of taxa, including a range of lizard species such as geckos, anoles, wall lizards and some iguana species (Goodyear & Lazell, 1994; Hite et al., 2008; Kwiatkowski & Sullivan, 2002; Mitchell, 1999; Ruiz de Infante Anton et al., 2013). The Schnabel Index has close similarities to the Lincoln-Petersen estimator, in fact it is considered to be a refinement of the Lincoln-Petersen estimator, it follows exactly the same assumptions and it is simply a weighted average of the collective Lincoln-Petersen estimates (Mares, Streilein, & Willig, 1981).

$$\widehat{N} = \frac{n_1 n_2}{m_2} \qquad \text{(Equation 2)}$$

$(\hat{N})$ Estimated population size

(n_1) Number of marked individuals captured in first capture session

(n_2) Number of marked individuals captured in second session

(m_2) Number of recaptures

The Lincoln-Petersen estimator (Equation 2) is a simple model, which is frequently used in population counts with capture-recapture methods (Lettink, 2012; Pollock et al., 2002). The main difference between the Schnabel and the Lincoln-Petersen estimate is that the Schnabel estimate uses data that has been collected over several days rather than the traditional Lincoln-Peterson estimate which uses data that is restricted to two sampling sessions. The first assumption of the Schnabel method is that the population is closed during the sampling period meaning there is no immigration or emigration. This is reasonable to assume given that the population of land iguanas is restricted to an island and individuals have limited opportunity to migrate between islands without human intervention. Another assumption is that the population is stable, with no births or deaths occurring during the study, this can be met as long as the study has taken place over a relatively short time period (Lettink, Armstrong, & Pryde, 2003; Seber, 1982; Sutherland, 2006). Data collection took place within approximately one year, with six sampling sessions per island occurring from August 2017 to July 2018, a short enough to meet this assumption. The final and most crucial assumption is that all individuals have an equal chance of being caught. This can be met as long as capture heterogeneity is to be acknowledged and accounted for, where capture heterogeneity refers to the realistic case that some individuals are less likely to be caught. For instance, age-biased dispersal can reduce the probability of smaller and younger individuals being caught (Lettink, Armstrong, & Pryde, 2003). In my study I have reduced the capture heterogeneity bias by excluding individuals whom do not meet the assumption of having equal probability of being caught. To achieve this, I limited the analysis to subadults and adults which were larger than 30 cm in Snout Vent Length (SVL). The small size of juveniles and their increased reclusive behaviour made them difficult to catch compared to their older counterparts and would violate the assumption of equal probability of capture.

Distance sampling

Similarly to Mark-recapture, distance surveys are a common method used for estimating animal abundance in a wide range of taxa including lizards (Ruiz de Infante Anton et al., 2013). The two main methods of distance sampling are line-transect and point-transect sampling, point transect sampling is primarily used for avian species. Line-transect surveys are generally used for ground dwelling animals. Distance sampling in comparison to Mark-recapture surveys is a non-invasive method as it is purely observational and does not require the animal to be captured (Buckland, 2001; Buckland et al., 2004), it is a far less stressful survey method for the animal. Line transect surveys are performed when the observer(s) move along the length of transect lines searching for the animal of interest on either side. Each time the animal is detected the perpendicular distance is recorded from the transect line with a laser range finder. Transect line length and location can be random or can be specifically chosen by the sampler. Line transect distance surveys accounts for imperfect detection of the desired subject animal, by modelling the detection probability with the distance from the transect line in the program distance (Buckland, 2001). Distance sampling, like Mark-recapture relies on certain key assumptions. These include that individuals are detected with full certainty at 0 m on the transect line, that the perpendicular distance from the individual and the transect line are measured accurately and that individuals do not move from their original location once detected. Some of these assumptions can be near impossible to meet due to the circumstance of the species or the survey site. For instance, small cryptic species or burrowing species can violate assumptions when not detected during the survey (Moore et al ., 2010). Thus, often one or more of the assumptions is allowed to be relaxed in contemporary distance surveys (Thomas et al., 2010). The first two assumptions can be met with a relative amount of certainty in our survey as land iguanas are large lizards and are fairly easy to detect in their environment, although they do burrow, we chose a time of the day that minimises the chances of an iguana being in its burrow. Accuracy of measurements is reliant on the laser range finder we used which records distances to the nearest meter. The latter assumption however is relaxed, as it is often that individuals do move from their original location due to being active in general or being startled.

I have chosen distance sampling as a method as it has proven to be an effective method for estimating lizard abundance. Additionally, the costs in terms of equipment, time and energy is much lower than the use of other methods. Our survey efforts included a total of 12 survey days, six on

Fernandina and six on North Seymour Island. Combined this is approximately 480 man hours of data collection spent on both islands. During data collection our field team was accompanied by two trained park rangers, who were assigned based on their level of experience with iguana handling and the study locations. In addition to assisting with navigation, the park rangers were responsible for capturing iguanas by hand and helped to train the field team on the appropriate techniques to minimise the potential harm caused during the capture and handling of iguanas.

Data collection

Mark-recapture surveys were conducted on North Seymour and Fernandina Island. Both islands had six sampling sessions. Searches and processing of iguanas occurred during daylight hours approximately between 5:30 am and 4:30 pm, with a 2 hour break generally from 12:00 pm to 2:00 pm where it was too hot to catch iguanas as their bodies were maximally active with the high temperatures in the middle of the day, thus they were harder to catch having increased agility and response. The early hours of the morning 5:30-7:30 am and afternoon 3:00-4:30 pm were the easiest time for catching the iguanas as they were torpid with the cooler surrounding temperature. The search was accompanied by two park rangers, my supervisor, student research assistants and myself. The team attempted to capture every individual that was observed within the study area. Upon capture one hand was placed behind the jaw of the iguana, while the other hand was placed firmly around the base of its tail. See Figure 1.5 for photographs of method described here. The captured specimen was then turned on its back exposing the underbelly, first it was checked for existing microchips with a with an Avid Minitracker 3 (AVID®). Then the snout vent length (SVL) and tail length measurements (TL) measurements were taken with a measuring tape to the nearest mm and recorded in centimetre (cm). The mass of the iguana was obtained by securing a noose rope under/behind the forelimbs of the iguana that was attached to a (Pesola®) spring scale and dropped, weight was recorded to the nearest 0.5g.

Sex was determined in the field based on a combination of characters routinely used by park rangers to distinguish between male and female land-iguanas. This includes: the number and size of spikes along the spinal crest, overall size, the size of the tail relative to SVL, and the size of femoral pits (Figure 3.3) It is generally accepted that sexual dimorphism in iguanas is associated with size, colouration and ornamentation. Males of many iguanid species have a larger body and head, with the

head possessing a thicker neck and pronounced dorsal spines or crest. Males also tend to have increased colour pigmentation and possess enlarged femoral pits, which are located on the inner thigh. Female iguanas on the contrary have smaller heads and bodies, with a duller colouration and noticeably smaller femoral pits (Mitchell & Tully, 2009). In the case of *C. subcristatus*, the males possess many features that a male iguana of any species would generally exhibit. Males were larger in size, with a thick neck and more pronounced dorsal crest (Switak, 1998), the yellow colour was more intense than that of females and the femoral pits were noticeably larger than the females. For juveniles and subadults the determination of sex was difficult as they do not yet show sexual dimorphism, since the sex of these individuals was unable to be determined accurately, it was recorded as unknown (Figure 3.3).

I classified land iguanas as either adult or juvenile/sub-adult on the basis of size and colouration. Smaller individuals (<30 cm SVL) that possessed dark tympanums (ears) and brown irises were classified as juveniles, or as subadults, if they were larger in size and exhibiting signs of sexual dimorphism. Iguanas that possessed white tympanums and more red irises were classified as adult.

A single12 mm Passive Integrated Transponder (PIT) tags (AVID®) were inserted under the skin of the inner left thigh. Left lateral profiles of the heads of each individual were photographed next to the specimen-specific transponder number using a Canon PowerShot ELPH 190 IS digital camera for identifying and cataloguing each captured individual. Prior to post-sampling release, iguanas were checked for any abnormalities including: missing claws, short or no tail, injuries and visible parasite load. A hand held GPS (Garmin® GPSmap 62S) was used to geo-reference the capture locations. Prior to releasing the iguana in the location it was found, white, non-toxic, water soluble paint was used to mark its dorsal surface to identify processed individuals. The handling time used for sampling and tagging was generally under 15 minutes from the time of capture.



Figure 3.3: Photographs of comparison of male and female land iguana and adult and juvenile land iguana.Top left shows male land iguana with pronounced femoral pits. Top right shows female land iguana with less pronounced femoral pits. This observation helps to determine the sex of a land iguana. Bottom left shows extreme size and colour variation of juvenile land iguana on left and adult male land iguana on the right. Bottom right shows juvenile land iguana. Photos: L. Ortiz-Catedral

Distance sampling

Distance sampling was conducted in two sample days only on North Seymour Island. We attempted a distance survey on Fernandina Island however this was a failed survey due to having very few sightings of iguanas (two). The uneven survey area, fissures and dense vegetation cover on Fernandina Island are not ideal conditions as it prevented the establishment of transect lines within the short time available for fieldwork, thus we abandoned this survey method for Fernandina Island.

Both the wet (December- May) and the dry (June- November) season were included: Sample days were; 14th of August 2017 and 1st of February 2018. The line transect survey method was used. In 2017 the survey was conducted between 10:00 am and 12:00 pm. In 2018 the survey was conducted between 7:00 am and 10:30 am. Iguanas start to emerge from their burrows once the sunrise has begun which is approximately around 6:00 am for both the wet and dry season, due to the equatorial position of the Galapagos archipelago (Jackson, 1993). By 7:00 am iguanas can fully emerge from their burrows especially if the weather conditions are optimal (sunny). We began the survey at 7:00am in August as

the weather conditions were optimal and the iguanas we already active. They are generally active and visible until the sun is starting to set where they retreat back to their burrows. The chosen survey times for both sample days were suitable for observing iguanas in their optimally active state. Transect lengths were decided based on our aim to cover the chosen area maximally, thus due to the arbitrary dimensions of the area Platforma, the transect lengths had differed in both sampling days. To avoid overlapping of transects and double detection of individuals we walked at least 50 metres away from the previous transect. Iguanas were not included and were difficult to detect if at 50 m. Transect directions were also altered. Distances were recorded with a laser range finder. Two observers were used including myself and one other experienced observer. Although one observer was assigned the job to handle the GPS and record notes both observers contributed to iguana observations to increase the precision of detection. It was made sure the pace along the transect was kept slow and, again to increase the accuracy of detection. If iguanas had moved from their original location post detection, then the original location was recorded as the detection location. The majority of iguanas were not startled during the survey, iguanas on North Seymour are relatively tame and docile towards humans from the distance, therefore there was minimal movement, aside from those found on the transect line, however this was reasonable to assume as 0 m. Iguanas at a perpendicular distance under 5 m were not detectable for the range finder, due to the detection distance being too small. For those iguanas the distances were estimated to the nearest metre by observation. Every detection was recorded along with the GPS location and number of individuals seen (some were in clusters). Observers had done their best to record every individual seen during the survey period.



Figure 3.4: Photograph showing the author and research assistant and park ranger on North Seymour Island just after conducting a distance survey. This photograph shows the conditions and landscape for distance surveys on North Seymour Island. Photo: L. Ortiz-Catedral

Data analysis

I used R version 3.4.1 for analysing Mark-recapture data. The total sampling area of our Mark-recapture survey On North Seymour is16 ha (Figure 3.8). On Fernandina Island the Mark-recapture survey area was 57 ha (Figure 3.9). I estimated the population of land iguanas on Fernandina Island using Mark-recapture estimates only.

For North Seymour Island both methods were used to form estimates. For distance sampling transect data, I used the program Distance version 7.1 (Thomas et al., 2010). The program accounts for imperfect detection of organisms by modelling the detection probability with the perpendicular distance from the target to the transect line (in this case, iguana). A one-observer transect method was used with the assumptions that iguanas on the transect line were detected with 100% certainty.

On North Seymour, we concentrated our distance sampling on the northern side of the island along a plateau known as the “Platforma” with a total area of 27 ha (Figure 3.11). For distance analysis two separate data sets were analysed from 2017 and 2018. I combined several different default functions offered by the program to obtain the best fit model. These included the key functions of half-normal,

uniform, hazard-rate and negative exponential and series expansions including cosine, simple polynomial and hermite polynomial. Additionally data was truncated at >20 m for every model in order to obtain a better fit, as detection probability is higher when closer to the transect line (Buckland, 2001; Thomas et al., 2010). These models and their respective estimates were evaluated for suitability with quantile-quantile plots, detection probability histogram and certain goodness of fit tests offered in the program. I used the Akaike Information Criterion (AIC) (Burnham & Anderson, 1992) to decide the model that best fit the data.

To compare population features of land iguanas between North Seymour and Fernandina Island I used Welch two-sample t-tests for unequal variances (McDonald, 2009). Data was tested for normality with the Shapiro-Wilks test, and showed that the data was normally distributed for all morphological variables ($P > 0.05$). A Chi-squared test was used for comparing proportions (McDonald, 2009). The variables compared between the populations are: SVL, mass and body condition as well as percentage of males, juveniles and injuries in iguana populations. Injuries include: cuts, scratches or bites found anywhere on the body, missing toes or parts of toes, and irregular tails. See figure 3.7 for injury examples. I excluded from analyses individuals that were under 30 cm SVL (n= 40) because these are less likely to be recaptured in subsequent sampling events.

Estimation of habitat availability

To estimate available habitat on Santiago I first quantified the available habitat on Fernandina Island. Previous research indicates locality of land iguanas on Fernandina Island and what vegetation type they occupy, however this has never been quantified. Iguanas do not occur across the entire island of Fernandina (64,200 ha), rather they are concentrated near the crater and the western flank of the volcano and around the crater rim which consists of sparse to thick vegetation. They are just not observed in areas with no vegetation (Werner, 1982, 1983).

With this information I quantified the area the of the island that corresponds to iguana habitat (sparse to thick vegetation) with satellite imagery from: http://institutodegeografia.org/vega-2/. This website details the vegetation type on each island and gives and the total amount of hectares per vegetation type on the island in a drop down list. For this I included deciduous forest (light green), evergreen forest (dark green) and deciduous shrubs (yellow) as land iguana habitat, see Figure 3.5.

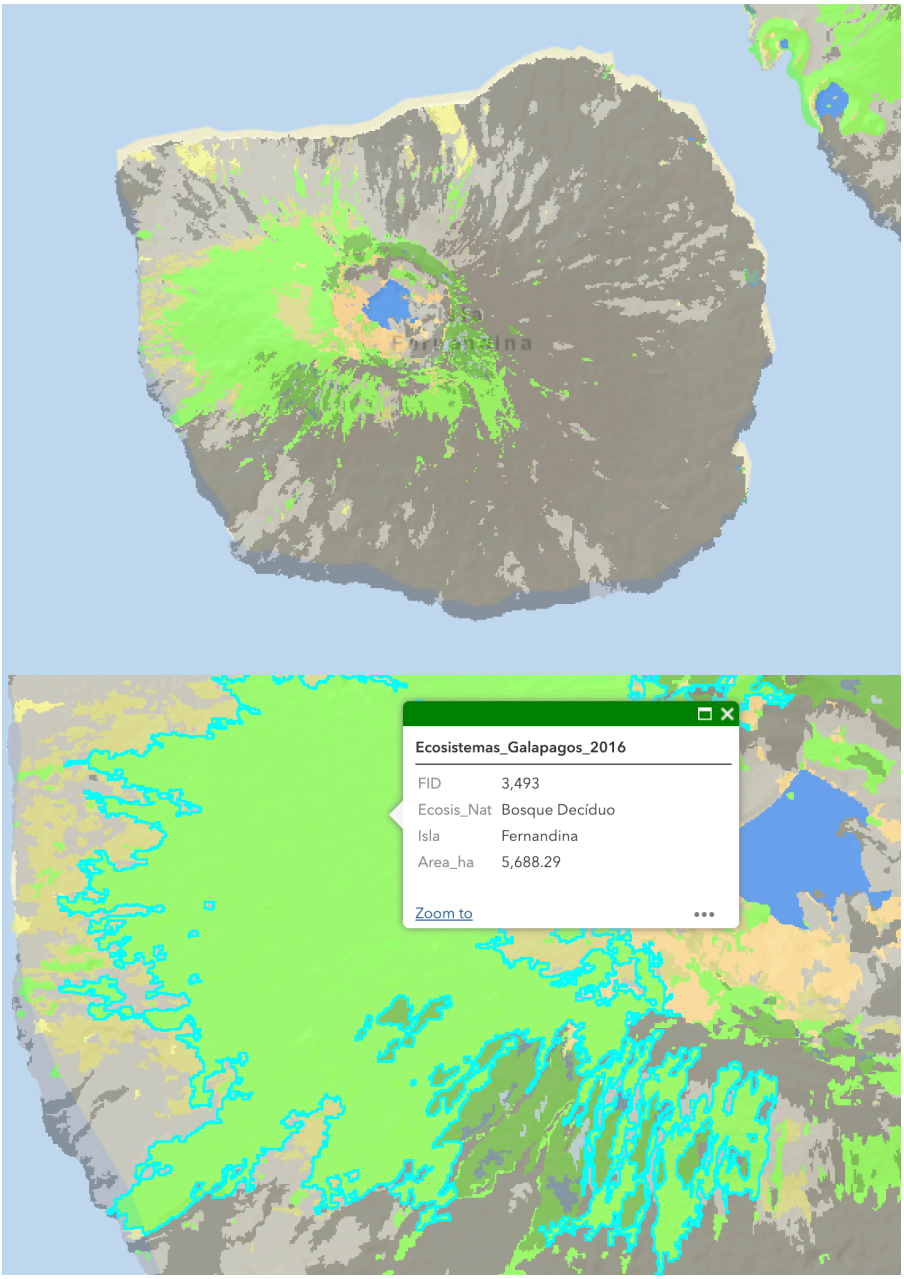


Figure 3.5: Image showing vegetation types on Fernandina Island. Top image shows Fernandina Island, light green, dark green and yellow represent land iguana habitat. Bottom Image shows light green area highlighted indicating the area size that consists of "Bosque Deciduo" i.e. deciduous forest at 5,688.29 ha. Sourced from: http://institutodegeografia.org/vega-2/

The same vegetation type was quantified on Santiago Island with the same method. See figure 3.6.

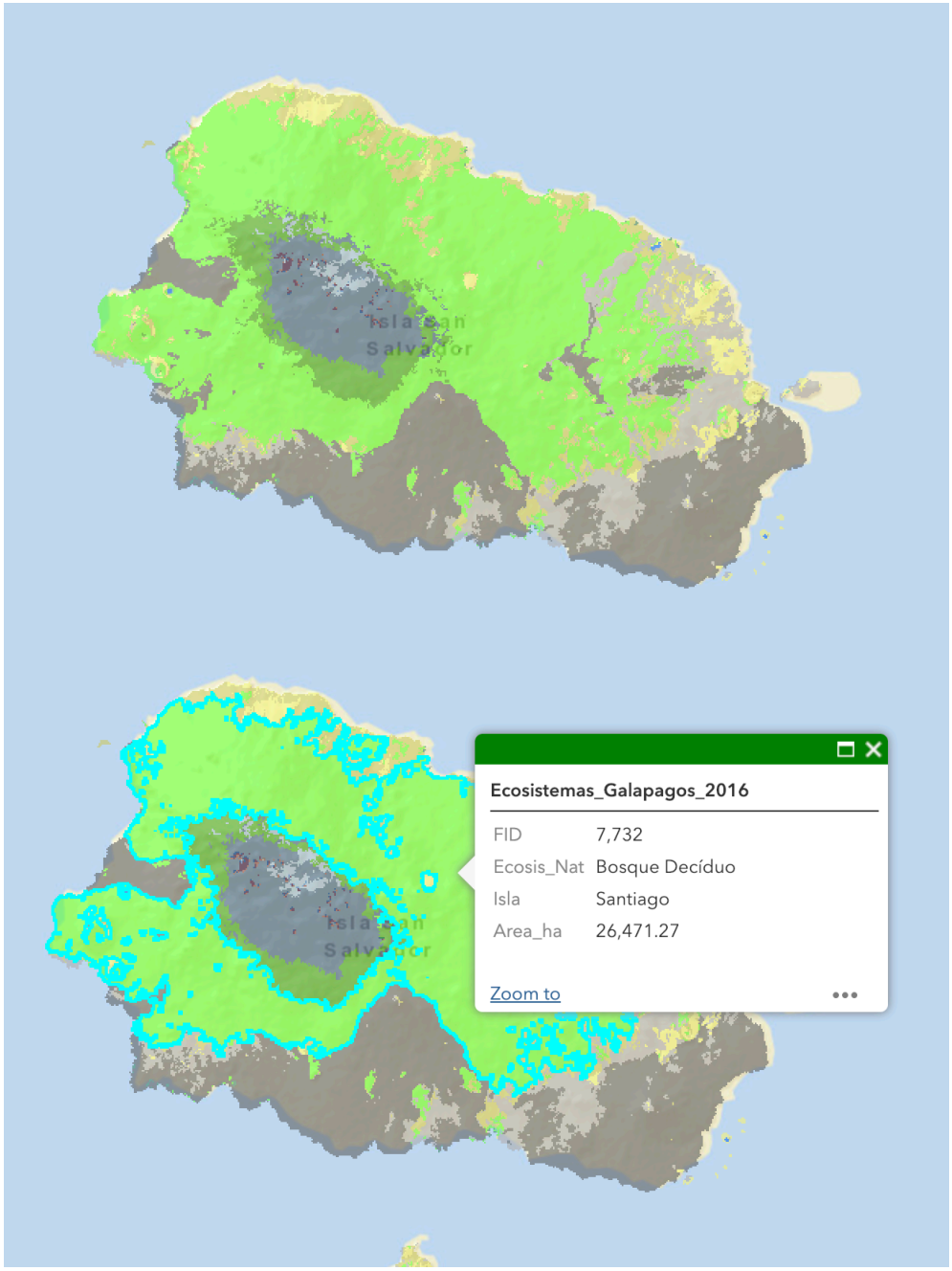


Figure 3.6: Image showing vegetation type on Santiago Island. Top shows Santiago Island; light green, dark green and yellow represent potential land iguana habitat. Bottom shows light green area highlighted indicating the area size that consists of “Bosque Deciduo” i.e. deciduous forest at 26,471.27 ha. Sourced from: http://institutodegeografia.org/vega-2/

RESULTS

Comparison of North Seymour and Fernandina land iguana population features

Differences in morphology:

The average SVL for iguanas from Fernandina was: males (n= 74) 51.74 cm and females (n= 69) 45.90 cm for females. Average SVL for iguanas from North Seymour was: males (n= 72) 52.87 cm; females (n= 42) 44.52 cm. There is no significant difference in SVL between the populations for both sexes ((t = 1, P = 0.14, males) (t = -1, P = 0.09, females)). There were significant differences in mass for both sexes between the land iguanas on Fernandina and the land iguanas on North Seymour ((t = -2, P < 0.05, males) (t = -4, P < 0.005, females)). With the average mass on Fernandina at: males (n= 74) 6.16 kg; females (n= 69) 4.16 kg. As for North Seymour the average mass was: males (n= 72) 5.62 kg; females (n=42) 3.42 kg. There were also significant differences in body condition indices between the Fernandina and North Seymour population of land iguanas ((t = -6, P < 0.005, males) (t = -3, P < 0.005, females)). Average body condition index on Fernandina was: males (n= 74) 44.60; females (n=69) 43.36. Average body condition index on North Seymour was: males (n= 72) 37.30; females (n= 42) 38.45.

Differences in injury proportions:

From comparing injury proportions between the two populations I would expect to get an insight on the amount of aggressive interactions in a population. These aggressive interactions could be explained by a number of environmental or social factors which will be discussed later in this chapter. Our results show a significantly higher proportion of injured individuals in the Fernandina population (22.24%) than the North Seymour population (9.72%) (X^2 = 9.9, df = 1, P < 0.05). I also found that over 70% of injured individuals were males for both populations, indicating intrasexual male aggression.



Figure 3.7: Photographs showing example of land iguana injuries. Left shows example of injury on head of land iguana, right shows example of injury of missing toes. Photos: Left: K. Kumar, Right: L. Ortiz-Catedral

Population structure

The analysis of male proportions was separated with the wet and dry season. The purpose of this separation was to see if there any significant differences in adult sex ratio between the seasons in the area surveyed. Breeding and laying season for Fernandina is known to occur in the April to May (wet season) and June (dry season) respectively (Werner, 1983). As for North Seymour the breeding and laying season is unknown. Separating the seasons to analyse adult sex ratio will allow us to pinpoint whether there are certain pressures the population is experiencing to exhibit their respective sex ratio. For the initial analysis we used data only from our study which began in 2017.

I found a male-biased adult sex ratio in both the populations in the wet season. A Chi-Squared test revealed significant differences in the proportion of males between the North Seymour population and Fernandina population of land iguanas, with North Seymour having 66.70% of males and Fernandina having 85.70% males ($X^2 = 8.7$, $df = 1$, $P < 0.05$). In the dry season, there was instead a female-biased adult sex ratio in Fernandina in the same sample area, as for North Seymour the male-biased sex ratio remained. Our analysis revealed a significantly higher proportion of males in North Seymour in the dry season (62.10%) than proportion of males (32.60%) in Fernandina in the dry season ($X^2 = 6.8$, $df = 1$, $P < 0.05$). There was no significant difference in male proportions between the seasons in the North Seymour population ($X^2 = 0.06$, $df = 1$, $P = 0.081$). However, there were highly significant differences in male proportion between the seasons in the Fernandina population ($X^2 = 51$, $df = 1$, $P < 0.0005$).

Juvenile proportion analysis:

This analysis reveals a significantly higher proportion of juveniles (28.65 %) in the North Seymour population than the proportion of juveniles (12.95 %) in the Fernandina population ($X^2 = 13.3$, $df = 1$, $P < 0.0005$). This includes both seasons.

Table 3.1: Summary of land iguana population features and analysis on Fernandina and North Seymour Island.

	Fernandina	North Seymour	P
n (males)	74	72	
n (females)	69	42	
Total	143	114	
SVL in cm (males)	51.74	52.87	P = 0.14
SVL in cm (females)	45.90	44.52	P = 0.09
Mass in kg (males)	6.16	5.62	P < 0.05*
Mass in kg (females)	4.16	3.42	P < 0.005*
Body condition (males)	44.60	37.30	P < 0.005*
Body condition (females)	43.36	38.45	P < 0.005*
Male percentage (%)	85.70 % (wet)	66.70 % (wet)	P < 0.05*
	32.60 % (dry) (n= 193)	62.10 % (dry) (n= 137)	P < 0.05*
Juvenile percentage (%)	12.95 % (n= 193)	28.65 % (n=185)	P < 0.0005*
Injury percentage (%)	22. 24%	9.72%	P < 0.05*

Note: in brackets wet referring to the wet season and dry is referring to the dry season
* indicates significance

Population estimates

Mark-recapture

A total of 185 iguanas were caught on North Seymour Island, of which 114 were including in the morphometric analysis and 163 included in the Mark-recapture population estimation analysis. On Fernandina a total of 193 iguanas were captured, with 143 individuals included in the morphological analyses and 176 individuals in the Mark-recapture population estimation analysis. Table 3.1 shows a summary of morphological variables between islands, sexes and age classes. Approximately 48 hours were spent capturing iguanas on North Seymour Island and 48 hours on Fernandina Island. For both populations the mean encounter rate I calculated was 7 iguanas per hour.

The Schnabel method estimated 10,857 iguanas per 16 hectares (95%CI: 2,315- 10,857) on North Seymour Island see figure 3.8. This equates to 679 iguanas per hectare, this estimate is based on 0 recaptures. Land iguanas are found across the entirety of the island. So if extrapolated to the entire island (190 ha), this gives us 129,010 land iguanas on North Seymour Island



Figure 3.8: Satellite image showing Mark-recapture survey area on North Seymour Island

Our survey site area size at Fernandina was 57 hectares, the Schnabel method estimated 1,745 iguanas per 57 hectares (95%CI: 869- 3816). This equates to 31 iguanas per hectare, this estimate is based on 6

recaptures. Iguanas do not occur across the entire island of Fernandina (64,200h), as mentioned earlier they occur mostly in vegetated areas around the crater rim and on the western flank (Werner, 1982, 1983). By using the vegetation maps provided by http://institutodegeografia.org/vega-2/, the total habitat area calculated for land iguanas on Fernandina is 7,105 hectares. Extrapolating our Schnabel estimate to the habitat area for land iguanas gives us an estimated 220,255 iguanas on Fernandina island.

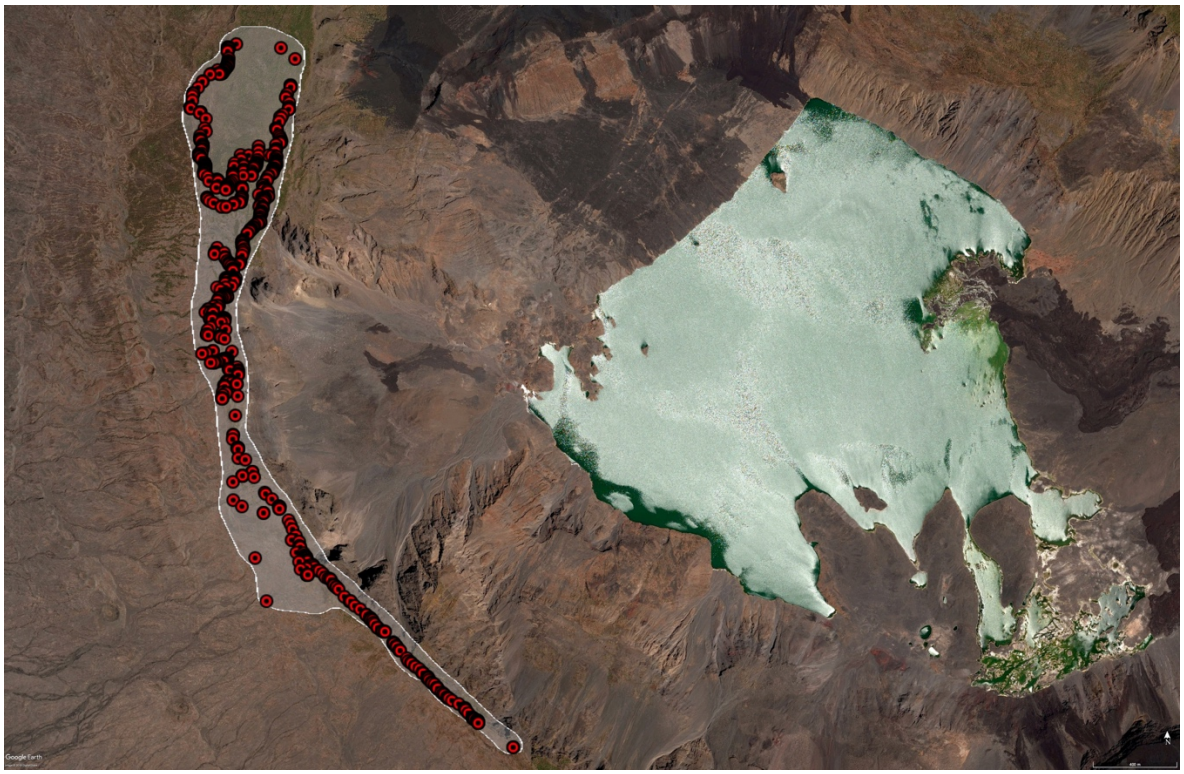


Figure 3.9: Map shows Mark-recapture survey area on Fernandina Island

For Fernandina island it took 4 field days to capture the first recapture; this is a total of 177 calendar days from the first Mark-recapture event 15th of January, 2018 till the first recapture captured on the10th of July, 2018.

Table 3.2: Summary of details on the recaptured land iguanas from Fernandina Island and estimated linear distance travelled between first capture and recapture

Code	Iguana ID	Date of 1st capture	Date of recapture	Days to recapture	Distance travelled (m)
A	9000260000577510	15/01/18	10/07/18	177	69.93
B	900026000577485	15/01/18	10/07/18	177	68.77
C	900026000577713	16/01/18	11/07/18	177	116.23
D	900026000577724	16/01/18	11/07/18	177	208.69
E	900026000577479	15/01/18	11/07/18	178	1249.48
F	900026000577757	15/01/18	11/07/18	178	1484.45

A Welch two sampled t-test showed non-significant changes in mass in recaptured individuals. The average change in body mass was 0.15 g more in the second capture (recapture). Recaptured individuals moved on average 532.93 m from initial capture location to recapture location. See figure 3.10 for capture locations.

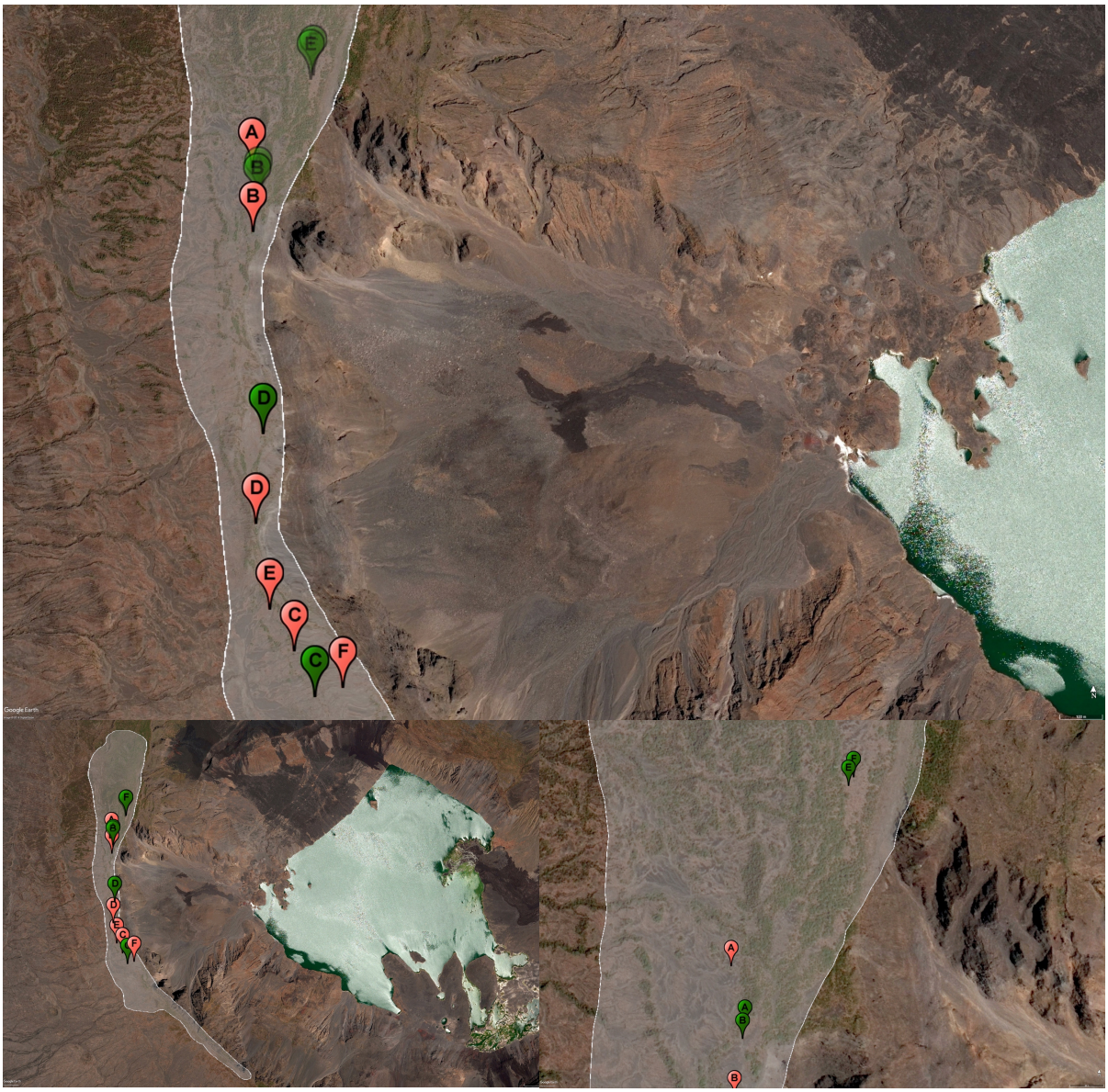


Figure 3.10: Satellite images showing recaptured land iguana locationsLocation of initial capture in green and recapture in red. Each letter corresponds to a land iguana, as shown in table 3.2. Note that A and B, as well as E and F are overlapped. Bottom left map shows survey area of Mark-recapture on the left side of the crater on Fernandina Island. Bottom right shows top map zoomed in to show clearly points A and B and points E and F (initial captures).

Distance sampling

2017 Survey:

The best fit model was the half-normal with a cosine series expansion. This data was truncated at >20m to obtain a better fit. This model ranked the highest in the program, as it had the lowest AIC score (379.85) There was a total of 66 observations. The estimate obtained is 44 ± 10 iguanas per hectare

(95% CI: 27-72). The Kolmogrov-Smirnov test: (D_n = 0.0918, P > 0.05) indicates a good fit to the data and the model had a low Coefficient of Variation (% CV) = 23.08. The encounter rate was calculated as 56.5m^{-1} and the detection probability as 43.5. Extrapolating this density estimate to the entire area we surveyed (Platforma) gives us 1,190 ± 275 iguanas per 27 hectares (95% CI: 732-19340. If extrapolated to the entire island of North Seymour that is a population estimate (N) of 8,376 ± 1,933 (95% CI: 5,153-13,613) iguanas in total.

2018 Survey:

For 2018 the best fit model was the hazard with a simple polynomial series expansion. This data was truncated at >20m as well to obtain a better fit. This model was chosen based on its green status, low AIC score (886.14) and the detection probability histogram, which showed a smooth decline with increasing distance from the transect line. There was a total of 154 observations, with slight increase in detections between 8 and 12 metres. The density estimate produced was 40 ± 12 iguanas per hectare (95% CI: 19-85). Similar to our 2017 density estimate. However, here the Kolmogrov-Smirnov test did not indicate as good a fit (D_n = 0.1494, P < 0.05). Nevertheless this was the best chosen model due to the relatively low AIC score as well as the histogram. The Coefficient of Variation (%CV) = 30 was acceptable. The encounter rate calculated was calculated at 64.7m^{-1} and the detection probability at 35.3. If extrapolated to the area of Platforma, it gives us 1077 ± 319 iguanas per 27 hectares (95% CI: 505-2298). If extrapolated to the entire island of North Seymour that is a population estimate (N) of 7581 ± 2246 (95% CI: 3,555-16,169) iguanas in total. The average estimate for the land iguana population on North Seymour Island including both 2017 and 2018 is 7,979 land iguanas.

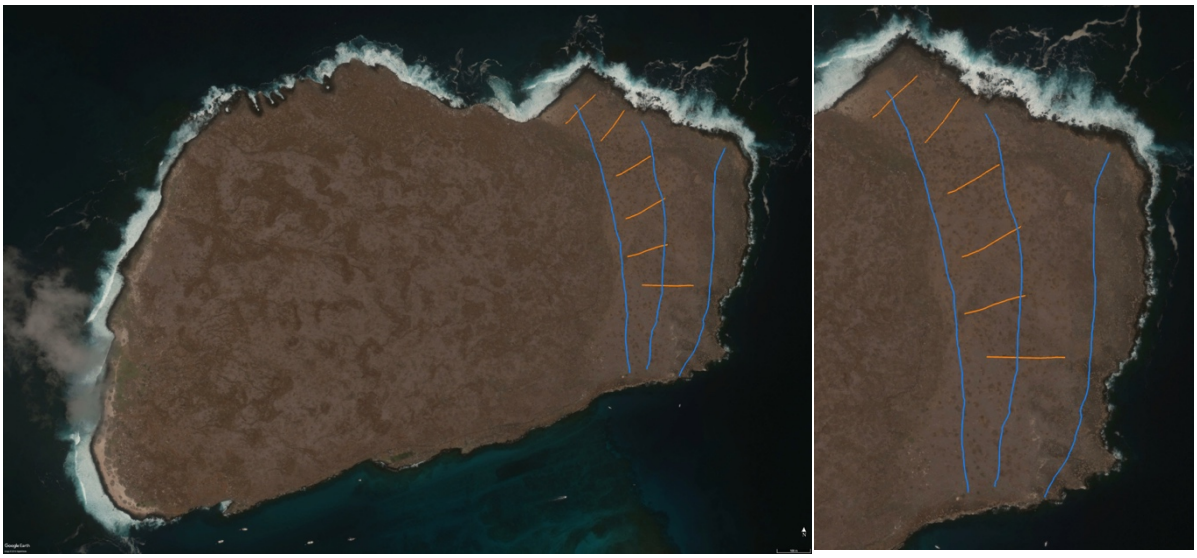


Figure 3.11: Aerial view showing distance transect lines for surveying iguanas on North Seymour Island. Transect lines span area Platforma. Orange lines indicate distance transect surveys from 2017, blue lines indicate distance transect surveys from 2018.

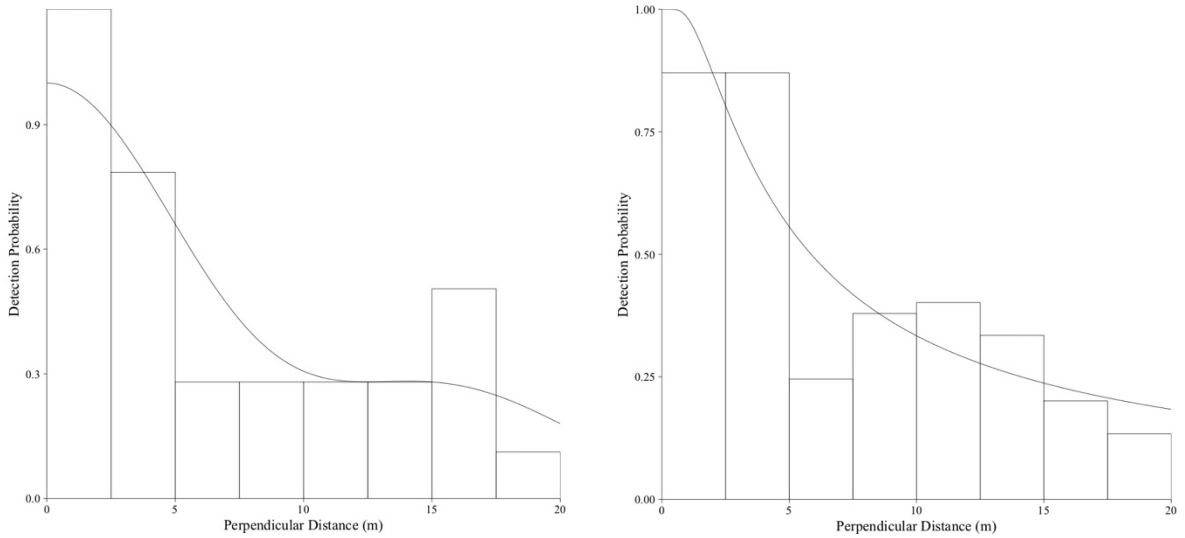


Figure 3.12: Detection functions for 2017 and 2018 distance surveys on North Seymour Island

Table 3.3: Population estimates for land iguanas on Fernandina and North Seymour Island, with the Mark-recapture method and the distance sampling method

	Mark-recapture estimate: iguanas per hectare (n)	Mark-recapture (N)	Distance density estimate: iguanas per hectare (n)	Distance estimate (N)
Fernandina	31	220,255	-	-
North Seymour	679	129,010	42*	7,979

* This is an average of the 2017 and 2018 estimate

Table 3.4: Comparison of Marquez et al. (2010) land iguana density estimates with this studies density estimates.

	Marquez et al. (2010)	This study
Fernandina (Mark-recapture)	1.88 (2006)*	31 (2018)*
North Seymour (Distance surveys)	22.78 (2007)*	42 (2018)*

* indicates year of sampling to produce the estimate

Estimates of available habitat for land iguanas on Santiago Island

Available land iguana habitat on Fernandina Island equates to 7,104 hectares. This includes deciduous forest, every green forest and deciduous shrubs. Available land iguana habitat covers 11.07% of Fernandina Island. By using the same principle, potential available land iguana habitat on Santiago Island is 27,042 hectares. This is 46.23% of Santiago Island.

Based on my land iguana density estimates on Fernandina Island, there are 31 iguanas per hectare of land iguana habitat, as calculated by the Schnabel method. This indicates that 31 iguanas per hectare is a standard land iguana density for Fernandina Island. If applied to potential land iguana habitat on Santiago Island, this gives us 838,302 iguanas in total. Under the following assumptions this number can be considered as the carrying capacity estimated for land iguanas on Santiago Island.

DISCUSSION

Introducing land iguanas to Santiago Island has been discussed and planned for since 2014 in the 2014 IUCN SSC Iguana Specialist Group Meeting (K. Campbell, pers. comm.). North Seymour Island was one of the main considerations for the source population for the translocation, partly due to the large population there. In preparation for this translocation, the population size and status of North Seymour land iguanas were necessary pieces of information. Additionally one of the key topics discussed in the Iguana Specialist Group Meeting was to investigate the degree of similarity in population parameters between Fernandina Island and North Seymour Island, whereby using Fernandina as a reference population as it is the only undisturbed population of land iguanas (Snell, Snell & Tracy, 1984) and consequently represents the "healthiest population of land iguanas. Additionally Fernandina Island shares a similar size and habitat of Santiago Island.

Past translocations of land iguanas have often been experimental (Perkins, 1930) or have had minimal planning, following population declines after feral dog attacks in the 1970s (Fabiani et al., 2011; Tzika et al., 2008). However, the tranlocation of land igaunas to Santiago Island had a different approach, hence questions such as how many land iguanas are needed to reach carrying capacity on Santiago and which island to used as reference population had arised, as North Seymour is not a suitable reference population owing to its density and size.

This chapter compares population parameters including age structure, adult sex ratio and also the proportion of injuries between the two populations Morphological characteristics were also compared, these are summarized in table 3.1. Recent population estimates were obtained for North Seymour and Fernandina Island. This analysis was carried out to fulfill the research needs that the IUCN SSC Iguana Specialist Group and Galapagos National Park delineated for the translocation.

Comparison of North Seymour and Fernandina land iguana population features

On average land iguanas from North Seymour Island are significantly lighter with lower body conditions when compared to land iguanas from Fernandina Island. the difference in SVL between the populations was non-significant. Interestingly, iguanas from North Seymour had a marginally larger average SVL (52.87 cm) than Fernandina Island (51.74 cm) despite having a significantly lower average mass and body condition than individuals from Fernandina. This suggests that land iguanas on both

islands share very similar lengths however their body condition differs significantly indicating that individuals from North Seymour Island are less fat with decreased body mass. Although, the average body condition index of 37.88 for male and female individuals on North Seymour is not considered as starved according to Costantini (2005) who indicates a body condition index of below 25 is a starved or drying individual. This number is significantly lower than the average body condition on Fernandina Island of 43.98 for males and females combined. Considering the iguanas share very similar size, in terms of body length between the islands, this suggests that iguanas on North Seymour are much thinner and have limited access or availability to food resources. This result is unsurprising for a number of reasons. As I have mentioned earlier, we have hypothesized and found in our distance surveys that the population of land iguanas on North Seymour Island is a dense population and may contain approximately 7,979 iguanas on a 190-hectare island. 190 hectares is a small area size and has a finite supply of food resources for iguanas. Carrying capacity of the North Seymour population of land iguanas may have reached and thus the island is not able to sustain the number of iguanas it currently holds, or it may not be able to support a growing population.

I infer the population on North Seymour Island is still continuing to grow as our analysis indicates a juvenile percentage of 28.65%. This represents a large proportion of the population which are part of recent population growth. This is a cause for concern as there are already signs of decreased body condition with the current population and this can be exacerbated with a growing population. It can be expected if the population is left to grow in the current state, food resources will be depleted resulting in further reductions in body mass and condition, consequently increasing their extinction risk. This could be a very real problem within the next decade. Therefore, translocating a portion of the land iguana population out of North Seymour Island will benefit the population there, by reducing intraspecific competition as well as reducing the grazing of *Opuntia* cacti (*Opuntia echios zacana*) which is hugely affected. We have observed several land iguanas grazing on a single *Opuntia* cactus. Additionally, many of the *Opuntia* appear to be in a degraded condition with rotting or black bases and minimal fresh green growth at the top of the cacti which the iguanas prefer and climb to. See figure 3.13. The species of cacti on North Seymour Island is a micro-endemic species, in that it only occurs on North Seymour Island (Helsen et al., 2009). A growing population of land iguanas poses a significant threat to the cactus population as may be depleted faster than their regeneration time. Furthermore, the

species of cacti here have evolved without grazers such as land iguanas prior to their introduction, therefore they have not evolved defenses against grazing animals unlike other *Opuntia* species in Galapagos, thus they are especially vulnerable to being overgrazed (Biggs, 1990).

Our results indicate a population of land iguanas that are in need of conservation interference, as there is potential for the entire ecosystem on North Seymour to be altered if the land iguana population is not controlled and left to grow. *Opuntia echios zacana* could be threatened with extinction and this can consequently affect other endemic species which depend on it, like the Galapagos finches (*Geospiza*), doves (*Zenaida*) and Mockingbirds (*Nesomimus*), which have all been reported to feed in *Opuntia* (Grant & Grant, 1981).



Figure 3.13: Examples of degraded *Opuntia echios zacana* surrounding by several land iguanas on North Seymour Island. Top photo: K. Kumar, bottom photo: L. Ortiz-Catedral

The dynamics of a population are often governed by recruitment, mortality rates and immigration. Information of sex ratios and age structure in a population are hugely important in determining the current status of the population and helps to predict the persistence of populations as determining factors of population recruitment and mortality which can significantly affect population growth rate (Andel & Aronson, 2012; Gillespie, 2010; Reichert et al., 2016). North Seymour had a male-biased adult sex ratio of 66.70 % in the wet season and 62.10% in the dry season. Fernandina had a major shift in adult sex ratio with a male bias sex ratio of 85.7% in the wet season and female biased sex ratio in the dry season (67.40 %). This surprised us at the time during the data collection period, we had only come across males in a certain area. As we walked further out of our study site, we noticed there were more females. Our primary study site thus was dominated by males.

However, in the dry season which runs from July to December we found instead a female-biased sex ratio with only 32.60 % of males in the same study site where we had previously found 85.70%. As for North Seymour the sex ratio did not vary significantly between the seasons and remained male-biased. The sex ratio variation from Fernandina is highly under documented. The extreme change in male to female dominated individuals in the same study site suggests there is an apparent segregation of habitats based on sex. Historical research on land iguanas from Fernandina Island states that all females in the western part of the island leave mating areas and migrate to the crater rim prior to July. The first two weeks of July were identified as the peak laying season. During the non-reproductive seasons the iguana population is reported to be distributed over a wider range (Burghardt & Rand, 1982). This research is seemingly consistent with our results. In the second week of July 2018 we found a female dominated study site near the crater rim. This coincides well with the peak laying season as mentioned before, thus explains the higher ratio of females. It is likely the males have moved to another habitat during the laying time. It is important to note that during both seasons the population of iguanas was concentrated at the summit near the crater rim, as opposed to the coastal area of Cape Douglas where we found only 13 iguanas out of the total 195 iguanas caught during our field seasons. Fernandina Island is the third largest (64,200 ha) in the Galapagos archipelago. Personal observations from our field seasons and past research show that the population of land iguanas is not uniformly spread throughout the island and rather appears to be in clusters with especially high densities of female land iguanas around at the summit (Werner, 1982, 1983). Thus, the highly male-biased sex ratio first

exhibited in the wet season as well as the female-biased sex ratio in the dry season cannot be treated as a true representation of the sex ratio of land iguana populations in Fernandina.

Sexual habitat segregation, during breeding and laying seasons, as well as the migration of females to the crater rim indicate ample space on Fernandina for land iguanas to show these patterns of occurrence. Analyzing previous unpublished data from Fernandina in 2004 till 2012 show a sex ratio with female bias (39.05% males), see table 2.2 in Chapter 2. This is perhaps the better representation of sex ratio of land iguanas from Fernandina. Additionally, Marquez at al. (2010) discovered a highly female-biased sex ratio on Fernandina Island with only 25.40% of the sample size representing males. Their data collection was at the crater summit on Fernandina and occurred in July 2006, this is during peak laying season when females migrate to the crater (Burghardt & Rand, 1982), which explains the highly female-bias. Their results corroborate with our results on adult sex-ratio on Fernandina in July (32.60 %). Therefore, this provides further support for sexual habitat segregation on Fernandina Island, indicating that land iguanas here are not competing for space.

Our results on adult sex ratio on North Seymour Island can be considered as a realistic representation of the sex ratio of iguanas. North Seymour Island is a small island (190 ha), during all our field days we saw that iguanas are distributed throughout the island in high densities, we had walked across the entire island from west to east and saw iguanas across the island. The island is heterogenous with mainly flat dessert area and sparse vegetation and some rocky areas and sandy beaches on the western and northern side of the island. Iguanas were seen in all of these habitat types. Our analysis shows a male-biased sex ratio for both the wet and dry season, both over 60% of males, these ratios did not significantly differ between the seasons. I found that 28.7% of the population we surveyed were juveniles. Our results along with our observations indicate that iguanas of all age and sex are distributed throughout the entirety of North Seymour Island. There is no observable segregation of males, females and juveniles, nor does there appear to be a preferred habitat type as iguanas are observed in all habitat types of North Seymour. Consequently, as opposed to Fernandina these results show possible space constraints for land iguanas on North Seymour, as we see that land iguanas prefer to use and occupy different habitats accordingly in Fernandina, however there is no option for selection in Seymour given the high density of land iguanas and limited space. To test the issue of space constraints for land iguanas on North Seymour I decided to conduct an analysis in the occurrence of injuries on land iguanas from

both populations as a comparison. I expected there to be a higher occurrence of injuries in the population from North Seymour with the assumption that spatial constraints are forcing male iguanas into closer proximity with smaller territories which can lead to increased aggressive interactions between males. However, our results contradicted our predictions. A higher occurrence of injuries (22.2%) was found on Fernandina Island as opposed to only 9.7% of iguanas injured on North Seymour Island. To explain the higher rate of injuries on Fernandina I assumed that the highly male-biased adult sex ratio of iguanas found in the wet season (December-May) caused an increase in intrasexual male aggression, additionally mating occurs during this time and males tend to have increased aggression, therefore our results are biased showing a high rate of injuries only as a result of the high proportion of males, 80% of injured individuals were found in concordance with the male-biased population in the wet season. For both populations over 70% of injured individuals were males. Male land iguanas are aggressive and territorial (Carpenter, 1969; Werner, 1982) a high occurrence of injuries within males is to be expected. As for the low rate of injuries on North Seymour Island, perhaps the territories are not small enough to fuel these aggressive interactions, also there is no congregation of males or females to bias our results. Further studies will need to be conducted to be able to make a sound conclusion, as the cause of injuries is not one hundred percent known.

As for our result of sex ratio of land iguanas on North Seymour Island, I conclude our results are a reasonable and fair representation of the true population ratio. Male-biased ratios in isolated populations can be a cause for concern and are recognized as a potential 'extinction vortex' whereby a sex ratio skewed towards males reduces breeding potential which rendering the population to be highly vulnerable to population declines and extinction (Alberts et al., 2004). This phenomenon has been mentioned in a number of studies (Galliard et al., 2005; Grayson et al., 2014; Palomares et al., 2012).

Lastly the juvenile ratio found on Fernandina Island (12.95%) was significantly lower than North Seymour Island (28.65%). I would suggest that the actual juvenile ratio on Fernandina Island is larger than 12.95%. This is mainly because Fernandina is a large island and juvenile iguanas are more difficult to find and may even occur in other parts of the island. During our data collection we were not specifically looking for juveniles, and so adults were the main age class caught, they are also much easier to catch and spot. North Seymour Island appears to have no habitat segregation between sexes or

age classes in land iguanas as we observed all sexes and age classes to be mixed throughout the island, thus we spotted juveniles more often, consequently making them easier to catch.

Population estimates

This chapter represents the first comprehensive population study on the population of iguanas from North Seymour Island after 27 years (Izurieta, 1991). Izurieta (1991) documented that the land iguana population was not increasing after their introduction to the island in the 1930s and suggested that it may in fact be a failed introduction. Since his study the land iguanas on North Seymour have experienced a population explosion, resulting in one of the most dense populations of land iguanas in the Galapagos (Marquez et al., 2010). Thus, this population has been subject to major changes since their introduction in 1932.

Much of the research concluded and suggest that Mark-recapture is a more reliable and unbiased method than distance sampling when estimating wild populations (Funk et al., 2003; Kacoliris, Berkunsky, & Williams, 2009; Ruiz de Infante Anton et al., 2013). In our study our Mark-recapture population estimates differed greatly from our line-transect distance surveys. Although I was only able to use and compare both ground survey methods only on North Seymour, this information is sufficient to gain an understanding on the appropriate survey methods to use on this island which will be discussed shortly. On average the population estimate for land iguanas on North Seymour yielded strikingly higher estimates of iguanas per hectare when compared to the Fernandina estimates. Suggesting that the population density of land iguanas is higher on North Seymour than on Fernandina, this was expected. However, the estimates for the Mark-capture study on North Seymour Island can be deemed as an unreliable estimate with 679 iguanas per hectare when compared with distance estimate of 42 iguanas per hectare. The Mark-recapture estimate is 16 times more than the distance estimate. With the six sampling days we had performed our study, we had no recaptures, thus having no recaptures greatly reduces the accuracy of the estimate with this survey method (Grimm, Gruber, & Henle, 2014) and explains the large disparity in population estimates between the two survey methods.

The present mark-recapture study on North Seymour and Fernandina took course over more than two days. Having multiple capture sessions increases the precision of estimates (Lettink, Armstrong, & Pryde, 2003). Despite having six capture sessions on North Seymour, which would

generally be considered as ample capture sessions to attain recapture data (Lettink, 2012), this was not sufficient. Other islands with recapture data include Isabela, South Plaza and Fernandina. Previous unpublished Mark-recapture studies on populations from these islands and my Mark-recapture study on Fernandina had all taken four days to recapture an individual. Some disadvantages lie in the violation of assumptions of this method. The accuracy and reliability of Mark-recapture and distance surveys are dependent upon the fulfilment of the given assumptions. Nevertheless, in practice these assumptions are likely to be unattainable, so contemporary surveys allow for relaxed assumptions. For Mark-recapture it is near impossible to fulfil the assumption of equal catchability in most wild populations. Populations are inherently heterogenous with individuals of varying size, age, behaviour and other features that can contribute to the variation in 'catchability' of an individual (Lettink, Armstrong, & Pryde, 2003; Pollock et al., 2002). Although our study accounted for this heterogeneity by only including individuals who were considered to have equal catchability (see earlier), this assumption still had room for violation. Iguanas that had been previously caught either recently (days) or in the previous year may have avoided being recaptured because of stress experienced due to the nature of this survey method. The individuals could have potentially sought refuge earlier than those who have never been captured or even avoided the location it was first captured altogether. This can be referred to as 'trap shy' behaviour (Lettink, Armstrong, & Pryde, 2003; Ruiz de Infante Anton et al., 2013; Sutherland, 2006). Although my study did not include the use of traps this could be analogous to hand capturing individuals. Behavioural response of marked individuals has been documented in a number of studies (Evans, 1951; Morris, 1955; O'Brien et al., 1985; Seber, 1970). Though, the behavioural response of marked iguanas in mark-recapture studies have not been investigated, it is not refutable, and may in fact have an effect on the probability of capturing a particular iguana depending on the individuals former capture history (Pollock et al., 2002).

Additionally, because our study had taken place within approximately a six-month period, the assumption for closed populations (no births or deaths) also has potential for violation, and it has been suggested in the past that an acceptable amount of days for sampling to occur in closed populations a minimum of 4-8 days to ensure precision of estimates (Lettink, 2012; Rees et al., 2011; Van Katwyk, 2014). Nonetheless I have chosen a closed population model as it is best suited to the data we sought to collect and was appropriate for the circumstance of our study. I consider this a short enough time to

conduct the analysis without having to radically jeopardise the significance of our results. However, the potential violation must be noted, as six months is enough time for these changes in the population to occur. It would be important to note however, that the hatching season on Fernandina occurs between September and December (Snell, Snell, & Tracy, 1984) and these months were not included in our study, consequently this should not have a significant effect on the population. Mortality in a population can occur at any time (Lettink, 2012), however we would have to assume none for the case of our study. As for North Seymour, the hatching season is unknown. Having no recaptures on North Seymour Island with the given number of days suggest that the population of land iguanas in North Seymour is extremely large, resulting in a severely low recapture probability. It would be highly unlikely to recapture an individual that has mixed in with a very large population.

The use of distance surveys as another method in our study is useful for comparing the two popular methods for estimating land iguana abundance despite having an invalid estimate for our Mark-recapture analysis on Platforma, North Seymour. Though we are not able to compare the estimates we are able to compare the use of the methods. Our abundance estimates from the distance analysis on Platforma, North Seymour can be considered as a fairly reliable and valid estimate. Distance surveys are advantageous for the reason being they are cost effective methods which require minimal time and effort in comparison to Mark-recapture. Although cost effective, the accuracy of the estimates has been questioned in previous literature. Distance survey estimates often underestimate abundance when compared to mark-recapture estimates. This disparity has been observed in previous studies and is not always unusual (Burton & Rivera-Milan, 2014; Rodda & Campbell, 2002; Smolensky & Fitzgerald, 2010). Low density estimates from distance surveys have been documented in previous iguana population studies. For instance, distance sampling has shown to underestimate densities of *Cyclura spp*. in the Caribbean, due to reason that this method relies on the assumption that all individuals in the study area are detected with certainty (Alberts, 2007; Burton & Rivera-Milan, 2014). Similarly, our density estimates were low for the likely cause being violation of assumptions as well as human error and possibly the time of day.

Although our density estimates were lower than the Mark-recapture estimates, the values produced in distance are more realistic and reliable as seen in the results. In this study we are confident that we met the assumption of certain detection if iguanas were on the transect line. However, the

assumption that that may have been violated and affected the accuracy of the results was likely the same for underestimates of density of *Cyclura spp.* It is almost certain that we in fact did not detect every iguana in our study area during the survey. When surveying we had only deviated from the transect line if we needed to walk around an obstacle, and the nature of the survey is to sample from a straight line. Thus, many iguanas may have not been detected outside of the transect line (0m). Because land iguanas are burrowing species, they may not all have been detected. Iguanas have been observed occasionally seeking shade in burrows during especially hot ours of the day. This can be between 11:00 am and 3:00 pm depending on the time of the year and the weather conditions on that day. In 2017 our surveys ran from 10:00 am and 12:00pm. During this time, it is possible that we have missed certain iguanas that were not available for detection because they were in their burrows. It is also possible some large sized *Opuntia* cactus may have intercepted potential detections, especially those that were further. In distance surveys observer variability is a common issue and appears in literature (Anderson et al., 1979). The observers current state whether fatigued or not or inexperienced in detecting iguanas as well as poor vision are all factors that could have contributed to increased undetected individuals. It is also possible that iguanas had sought refuge or cover in a burrow, rock or behind a tree if they had observed us approaching. However, we had taken care to minimize this affect as mentioned earlier. We had also made sure the area being surveyed was free of any disturbance prior to the sampling session, which allowed for the iguanas to be in their natural state. Given all these possibilities for missed detection does not necessarily indicate that our survey will fail to give reliable results. Distance surveys are realistic to biological situations in that in is inevitable that many individuals are left undetected and this is fully allowed for, given that this applies to individuals outside of the transect line and not at distance 0 m.

Our abundance estimates from the distance analysis for 2017 and 2018 on North Seymour land iguanas were slightly lower than expected, however I consider them to be a reliable indicator of the true population density. During the distance surveys we had collected data on the demographics of the surrounded land iguanas population given in table 3.1. Considering the percentage of juvenile iguanas is 28.65 % I would consider our distance observations of iguanas to also entail a similar proportion of juvenile iguanas. The size of the iguana can have an effect on our ability to detect with precision every iguana in our survey area. Thus, it is fair to consider the fact that perhaps at least 20%~ of iguanas were

left undetected owing to their small size and this could have ultimately led to a lower density estimate on completion of the analysis

The distance survey estimates were significantly more reliable than our Mark-recapture estimates, however, with the limitations stated previously for distance sampling I conclude that our results are possibly a mild underestimate, as we had expected a higher density of iguanas per hectare, and it is inevitable some iguanas were missed during our survey. There is huge potential for significantly improving distance results with minimal increases in sampling effort, such as repeated surveys on the transect lines either on the same day (different times of day) as well as further sampling sessions on added days as well increasing the amount of transect lines. Due to our time constraints we were only able to perform surveys once on a given transect line. For this same reason we were restricted to the amount of transect lines we could survey, furthermore the rest of the time granted was dedicated to Mark-recapture surveys.

Further limitations on both populations estimate methods on North Seymour include site access to certain areas of North Seymour Island. Iguanas appear to be distributed across the entirety of the island. Given the time and other constraints we had, we were only able to survey a portion of the island (14.2 %). The tourist site was also off limits owing to GNP rules and research site restrictions. Additionally, many parts of the island are highly rocky and unstable, conducting either of the surveys would be difficult and dangerous. Surveying Platforma was most convenient and suitable. It has an advantage of extremely flat ground and limited arid vegetation thus visibility here is better than other parts of the island. Given extra funding, time and park ranger service the abundance of iguanas on North Seymour can be estimated with increased precision. Sites not included in our study could potentially be included and stratified in our distance analysis.

The population estimates are an update since Marquez et al's (2010) study. Table 3.4 shows a comparison to our population estimates. Our density estimates based on the Mark-recapture method for Fernandina of 31 iguanas per hectare is over 15 times more than the mark-recapture density estimates of Marquez et al. (2010) (1.88 iguanas per hectare). It is unknown how many recaptures were in their study. The discrepancy between my estimates and Marquez et al's (2010) estimate can be explained by the highly variable sampling sights. Their study was focused on the northern part of the crater whereas ours was on the western side of the crater which is known for containing a higher density of iguanas.

Thus furthermore, our density estimate of 31 iguanas per hectare does not necessarily apply across every land iguana habitat patch. Certain areas do exhibit increased land iguana density. During sampling on Fernandina island I noticed land iguanas were particularly dense at the crater edge. Whereas the density decreases even as far as 300 m from the crater edge.

These findings suggest that one can expect to find major discrepancies in population estimates on large islands with large populations. Additionally, the actual habitat of land iguanas on Fernandina has never been quantified. Our study is the first to give an estimate of the habitat range of land iguanas from Fernandina. Further studies on land iguana range on Fernandina must be conducted in order to yield a reliable estimate. Furthermore if reliable population estimates from Fernandina are to be obtained, intensive sampling effort must be invested, given the large area size of the island as well as the large population (Lee et al., 2014). The population estimates for North Seymour land iguanas from Marquez et al. (2010) study of 4,100 iguanas are also based on a distance survey. The estimate was obtained in a single sampling session (1 day) with a total of two transects. The of the location of the transects in Marquez et al's (2010) study is not given. Our distance population estimates for North Seymour Island of 7,979 iguanas is given 11 years after Marquez et al. (2010). It is likely that the population of land iguanas from North Seymour has exhibited in increasing population trend since their introduction as well as since Marquez et al's (2010) study. Consequently, I infer our population estimates are a more reliable indicator of the real population of land iguanas on North Seymour as they are current and our study had two sampling sessions with a total of nine transects.

Habitat availability and carrying capacity on Santiago Island

Based on habitat occupancy of land iguanas recorded on Fernandina Island (Werner, 1982, 1983). I was able to estimate the potential available habitat on Santiago Island in the same manner as described in the methods section. The potential habitat area that land iguanas could occupy on Santiago is estimated at 27,042 hectares. Although iguanas do not occur in uniform density across their habitat on Fernandina Island this gives an indication of the potential space they could occupy as well as how many individuals could occupy the entire calculated land iguana habitat. Our Mark-recapture estimates on Fernandina Island reveals a density of 31 iguanas per hectare. Fernandina Island holds the only undisturbed population of land iguanas, therefore this density estimate can be considered as an

acceptable density for land iguanas in an optimal habitat. When extrapolating this density to the potential available land iguana habitat on Santiago Island we can expect 838,313 iguanas to colonize Santiago Island. This estimate is referring to the potential carrying capacity of the island. This part of the analysis was explored only briefly due to limited funding for the project, however exploring habitat availability and colonization of land iguanas on Santiago Island should be a key research priority. This, is discussed further in Chapter 5.

To conclude and address our aims for this chapter regarding estimating population size, the Mark-recapture estimates for North Seymour land iguanas is not reliable, our Fernandina population estimate is a more realistic estimate, however we cannot count it as an accurate estimate as such but only an inference. Conversely, our distance estimates were considerably more reliable. Though, no study in practice is ever able to satisfy the assumptions completely, this is widely known and thus it is accepted to compromise on precision (Begon, 1979). Having no recaptures greatly reduces the precision of these estimates. However, I believe this method would still be a suitable method to use for estimating the population size for land iguanas on North Seymour Island, given sufficient funding and increase sampling effort. More sampling sessions would eventually yield recaptures, and we would expect to see an increase in reliability of estimates. The results for Fernandina suggest that the Mark-recapture method has potential to yield reliable estimates as long as we have recaptures. Increasing the time and sampling effort would increase the amount of recaptures and our sample size, this is a universally known way to increase precision of estimates in population surveys (Begon, 1979; Lettink, 2012; Lettink, Armstrong, & Pryde, 2003). As demanding as Mark-recapture surveys are, their ability to produce reliable estimates has been proven in the past, this warrants the costly investment of this method. However, the study site must be suited to the method.

With the outcome of this analysis, I suggest distance surveys are the appropriate survey method to use, especially for smaller islands/ populations. This includes North Seymour, Baltra, South Plaza, Cerro Dragon (Santa Cruz) and Venecia, thus distance surveys are ample to survey many populations of land iguanas in the Galapagos. Distance surveys are non-costly and require much less effort and investment than mark-recapture surveys, additionally they have proven to provide a reliable estimate, based on the outcome of our analysis. Mark-recapture may be more suitable and yield more reliable estimates if the population size was not overly large. As both Fernandina and North Seymour hold a

very large population of land iguanas, the use of mark-recapture methods to obtain population estimates is colossal effort, therefore I do not suggest using mark-recapture to survey large populations of land iguanas unless there is ample funding, time and efforts invested for the long-term. As for our estimates of carrying capacity on Santiago Island (838, 313), these extrapolations were based on preliminary data and limited sampling effort due to funding constraints as well limited spatial data on the remote location of Santiago. Thus this only indicates the potential population size that land iguanas can reach on Santiago Island. This will need corroboration with further research.

My results indicate a highly dense population of land iguanas on North Seymour Island and a diverse representation of age classes. It is this high density that makes Mark-recapture estimates so unreliable, because the inter-annual (i.e. 2017-2018) effort required to begin obtaining recaptures is much greater than on Fernandina Island. North Seymour shows a proportion of juveniles more than twice higher than on Fernandina, and lower proportion of injuries. What these findings mean in the context of a translocation of the species to Santiago Island, is that sourcing thousands of individuals from a single site with recruitment of juveniles is feasible due to the high densities estimated. Further, the lower proportion of injuries is favourable as individuals are "used" to co-existing in crammed conditions, an unavoidable situation during the transfer of numerous individuals for translocation. The differences in population parameters on North Seymour and Fernandina Island uncover the current state of land iguana on North Seymour Island. The key finding of lower body condition and high density supports the decision of North Seymour Island to be one of the main sources for the translocation of land iguanas.

CHAPTER 4

A Review of the IUCN Threat Category for Land Iguanas



Figure 4.1: A Galapagos land iguana (*Conolophus subcristatus*) on North Seymour Island. Photo: L. Ortiz-Catedral

ABSTRACT

The IUCN Red List is a widely used conservation tool to guide the management of endangered or threatened species. Therefore having an up-to-date Red list assessment is beneficial for effectively managing the species of interest. The current IUCN Red List assessment for land iguanas has not been reviewed since 1996. Land iguanas are currently classified as 'Vulnerable' under the IUCN Red List and the information provided in the assessment is lacking any information on the species. This chapter aims to update the conservation status of the land iguana and reassess the threat category given. The bulk of the chapter is presented as a completed IUCN Red List Assessment Questionnaire provided by the IUCN. The questionnaire covers several aspects on the species including the distribution, ecology, population specifics, threats and conservation actions. Key findings include the potential for projected population declines of land iguanas in the next 100 years in populations that are exposed to cat predation. Based on the projected population declines of land iguanas as well as its limited geographic range I conclude with reinstating the threat category of 'Vulnerable'.

INTRODUCTION

The IUCN Red List was first developed in 1964 by the International Union for Conservation of Nature for the purpose of conserving threatened species based on a given criteria of threat. It is universally recognised as the most comprehensive inventory of biological species to date ("The IUCN Red List of Threatened Species," 2017). The information the Red List provides is often used as a basis for any conservation decisions for species worldwide as well as guiding research focus initiatives (Jarić et al., 2017). The IUCN Red List system includes nine categories that range from “Not evaluated” (NE) to “Extinct” (E). The intermediate values starting from above NE include; Data deficient (DD), Least concern (LC), Near threatened (NT), Vulnerable (VU), Endangered (EN), Critically endangered (CR) and Extinct in the wild (EW). The latter having the highest risk of extinction. These threat categories are used as the fundamental guide for prioritizing conservation investment within species. Thus, higher threat categories such as DD and CR species are identified as research priorities and where conservation efforts are generally weighted towards (Jarić et al., 2017; Rodrigues et al., 2006). Because conservation efforts and research initiatives are largely influenced by the IUCN Red List threat category, it is therefore imperative to have a threat category that correctly reflects the species and also one that is up-to-date with current information. The Red List contains data on over 79,000 species, however, the amount of data on species is skewed within groups. Larger, charismatic species have generally received significantly higher research focus and conservation efforts in the past, these groups include mammals such as tigers (*Panthera tigris*) and panda bears (*Ailuropoda melanoleuca*) (Ducarme, Luque & Courchamp, 2013). Consequently, groups such as amphibians, fishes and reptiles are less described with less information (Clark & May, 2002).Thus, this can result in negligence of conservation attention.

The IUCN Red List entry for Galapagos land iguanas (*Conolophus subcristatus*) lacks general information, and is in need of updating. The Red List category it is assigned is Vulnerable. This was first assigned in 1986 and continued to be Vulnerable until 1996. Since then, there have been no updates on this species. There is no information on its geographic range within Ecuador, there is also nothing on habitat and ecology and conservation actions, aside from “It is listed on the CITES Appendix II” ("The IUCN Red List of Threatened Species," 2017). There have been some developments in the IUCN Red List criteria and categories since 1996, when the land iguana was last reviewed. In 2001 the categories were redefined (Proudlove, 2004). Thus, along with the outdated land iguana Red List entry,

the conservation status of "Vulnerable" may not reflect its true status. There is no information given to support the assigned category, and there have been several research advancements on *C. subcristatus* since 1996.

In this chapter I will present the relevant information needed to evaluate the IUCN Red List threat category for the Galapagos land iguana, in case a change is required. All information used to evaluate the threat category is based on a literature review of land iguanas as well as the outcome of some analysis from this thesis. This will update the current land iguana Red List entry with up-to-date information that has been published in the last two decades on *C. subcristatus.* To have received the threat category of Vulnerable in 1996, the land iguana would have been reviewed under the framework of version 2.3 of the IUCN Red List Categories prepared by the IUCN Species Survival Commission, published in 1994 (IUCN, 1994). In this chapter I will be using the most recent version of 2001 IUCN Red List Categories and Criteria version 3.1 as well as Guidelines for Using the IUCN Red List Categories and Criteria Version 13 (IUCN, 2017) to evaluate the threat status of *C. subcristatus*. Currently the IUCN requires Red List Assessments to be entered and managed in the IUCN Species Information Service (SIS). The SIS is the central database used by the IUCN and it is now used as the main publishing platform for Red List assessments. However, the SIS is restricted to certain users who are involved in IUCN Red List assessment projects currently. Therefore instead, I am using the Red List Assessment Questionnaire template that is provided by IUCN, as this was previously used as the means to update or create a Red List entry prior to the SIS system. This chapter is presented as a completed questionnaire provided by the IUCN. All information is relevant and all sections in the questionnaire are applicable to the current IUCN Red List entry requirements. The information in this chapter is the most up-to-date on the land iguana and it will be presented to the Ecuadorian members of the SIS for review and incorporation on the IUCN Red List Assessment of the Galapagos land iguana.

RED LIST ASSESSMENT QUESTIONNAIRE

1. SCIENTIFIC AND COMMON NAMES.

The IUCN Red List generally focuses at the species level. Subspecies, plant varieties, and subpopulations (as defined in the IUCN Red List Categories and Criteria. Version 3.1) may also be assessed and will be considered for inclusion in the IUCN Red List only if the species-level assessment is also available. Hybrids will not be considered for inclusion in the IUCN Red List. For currently undescribed species, please refer to the rules outlined in section 2.1 of current version of the *Guidelines for Using the IUCN Red List Categories and Criteria*.

1a. Scientific name (including authority details):

Conolophus subcristatus (Gray, 1831)

1b. Synonym/s (if there has been a taxonomic change in the last 5 years or if widely used):

Amblyrhynchus subcristatus (Gray, 1831)

1c. English Common Name (if known):

Galapagos Land Iguana

1d. Other Common Names (if known and state language):

Common Land Iguana Yellow Land Iguana Iguane terrestre des Galápagos (French) Iguana Terrestre De Las Galápagos (Spanish)

2. HIGHER TAXONOMY

Note that the IUCN Red List does not record sub-families, sub-orders, etc. Only the taxonomic levels requested below should be provided. A taxonomic notes field is also provided to allow further details about taxonomy to be recorded – see section 4a.

2a. Kingdom	**2b. Phylum**	**2c. Class**
ANIMALIA	CHORDATA	REPTILIA

2c. Order	**2b. Family**
SQUAMATA	IGUANIDAE

3. COUNTRY, SUBCOUNTRY AND MARINE AREA OCCURRENCES

Provide a list of the **countries** and **subcountry units** (e.g., states, provinces, etc.) in which this taxon occurs. For marine taxa, also record names of FAO fisheries areas and (optional) Large Marine Ecosystems (LME).

Presence: For each country, subcountry or marine area, please record whether this taxon is extant, extinct, possibly extinct, or presence uncertain.

Origin: For each country, subcountry or marine area, please record whether this taxon is native, reintroduced, introduced, vagrant, or origin uncertain.

Note: A distribution map showing the extent of occurrence **MUST** also be attached.

See the current version of the *Guidelines for Using the IUCN Red List Categories and Criteria* for the IUCN definition of "extent of occurrence".

3a. Countries			**3b. Subcountry units** (if known)		
Country name	**Presence**	**Origin**	**Subcountry unit name**	**Presence**	**Origin**
Ecuador	Extant	Native	Fernandina Island	Extant	Native
			Isabela Island	Extant	Native
			Baltra Island	Extant	Reintroduced
			South Plaza Island	Extant	Native
			North Seymour Island	Extant	Introduced
			Santa Cruz Island	Extant	Native
			Venecia Islet (Santa Cruz Island)	Extant	Introduced
			Santiago Island	Extant	Reintroduced
			Rabida Island	Extinct	Native

3c. Marine Areas		
FAO area name or LME	**Presence**	**Origin**
[N/A]		

4. TEXT DOCUMENTATION

Provide a short narrative for each of the topics below to support the information used for the Red List assessment in section 5 and to complement and provide more detail for the Classification Scheme codes for habitats, threats, stresses and conservation actions recorded in Annex 1 (use additional sheets if required).

Please avoid using one-word answers in this section; the Red List assessment should be treated in the same way as a scientific paper, where the information is presented as clearly as possible for the reader, and all references used as cited within the text.

4a. Taxonomic Notes

Record any recent taxonomic changes or current taxonomic doubts or debates about the validity or identity of the taxon.

This species was assigned to the genus *Amblyrhynchus* and the species *subcristatus* by (Gray, 1831).

Previous taxonomic changes include:

Amblyrhynchus Demarlii (Duméril, Bibron, & Duméril, 1839)
Hypsilophus (Conolophus) Demarlii (Fitzinger, 1843)
Trachycephalus subcristatus (Gray, 1845)
Conolophus subcristatus (Steindachner, 1876)

Subspecies include (now considered separate species):
Conolophus pallidus (Heller, 1903)
Conolophus marthae (Gentile & Snell, 2009)

4b. Distribution

Provide a summary of the current information available about the taxon's geographic range. Include a mention of important sites for this taxon.

The Galapagos land iguana is endemic to the Galapagos Islands (Ecuador), it is native and extant to five islands and introduced to one. The islands where the species occurs nowadays include: Fernandina, Isabela, Santa Cruz, South Plaza, Baltra and North Seymour. There are at least 11 known populations of land iguanas on these islands (Marquez et al., 2010). Land iguanas are found in arid to semi-vegetated regions of the island (Jackson, 1993; Werner, 1983). Iguanas are distributed across the entirety of South Plaza, Baltra and North Seymour Island. South Plaza and North Seymour Island are both small islands and hold a dense population of land iguanas (Snell, Snell, & Tracy, 1984; Tzika et al., 2008). Fernandina, Isabela and Santa Cruz Island are all larger islands, where there is more than one population present and currently there are various degrees of separation between populations. On Fernandina, land iguanas are dense on the western flank of the island's crater (Werner, 1983). Isabela is the largest island and contains at least five populations of land iguanas (Tzika et al., 2008). The entire distribution of land iguanas on Isabela Island is unknown. Santa Cruz Island holds three populations of land iguanas, with one of the populations on an islet called Venecia, introduced in 1976 (Cayot, 2008; Fabiani et al., 2011).

Historical populations of land iguanas occurred on Santiago Island, Rabida islet and across Santa Cruz Island. However, since the 17th century, the introduction of mammals had caused the extinction of iguanas on Santiago Island, and large portions of Santa Cruz Island (Cruz et al., 2005). The cause of extinction of the population of land iguanas on Rabida islet is unknown but appears unrelated to human activities (Steadman & Zousmer, 1988).

4c. Population

Provide a summary of the information available for size and trend of the global population. Information about sizes and trends of subpopulations or trends in particular regions of the taxon's range can also be included in this section. If no quantitative information on population sizes or trends is available, please record whether the species is common, abundant, rare, etc. If there really is no information at all about the population, please note this.

The Galapagos land iguana is generally a common species on the islands they occupy. It is the most common iguana of the *Conolophus* genus. Current population estimates and trends are unknown. Previous population estimates for the entire population of land iguanas concluded that there are over 10,000 land iguanas across 11 confirmed populations in the Galapagos archipelago, with the largest populations found on Isabela Island, Fernandina Island and North Seymour Island (Marquez et al., 2010). Land iguanas are especially abundant and found in dense populations on North Seymour Island and South Plaza Island. The populations of land iguanas on Isabela and Santa Cruz Island can be considered as fragmented as they do not occur across the entire Island, only in clusters (Tzika et al., 2008). Land iguanas cannot swim and have never been reported to disperse outside their island of occupancy, thus they have limited dispersal abilities only between islands.

The most recent population estimates of land iguanas on North Seymour Island and Fernandina Island are given by (Kumar, Ortiz-Catedral, & Llerena, 2019). Kumar, Ortiz-Catedral, & Llerena (2019) estimated 7,979 land iguanas on North Seymour Island and 220,255 iguanas on Fernandina Island, note the latter estimate on Fernandina Island could be an overestimate due to limitations in the Mark-recapture study. The, estimate on North Seymour is considered an acceptable estimate and is based on distance line-transect surveys. Overall population estimates for land iguanas across all islands remains unknown. Current population trends are also unknown, it is however likely the population of land iguanas in North Seymour Island has reached carrying capacity. Previous recorded population declines have occurred last in the 1970s due to feral dog attacks on Santa Cruz and Isabela Island (Fabiani et al., 2011; Tzika et al., 2008). It is likely some populations of land iguanas are currently in decline. The table provided below presents current hypothesized trends for different populations of land iguanas based on each island

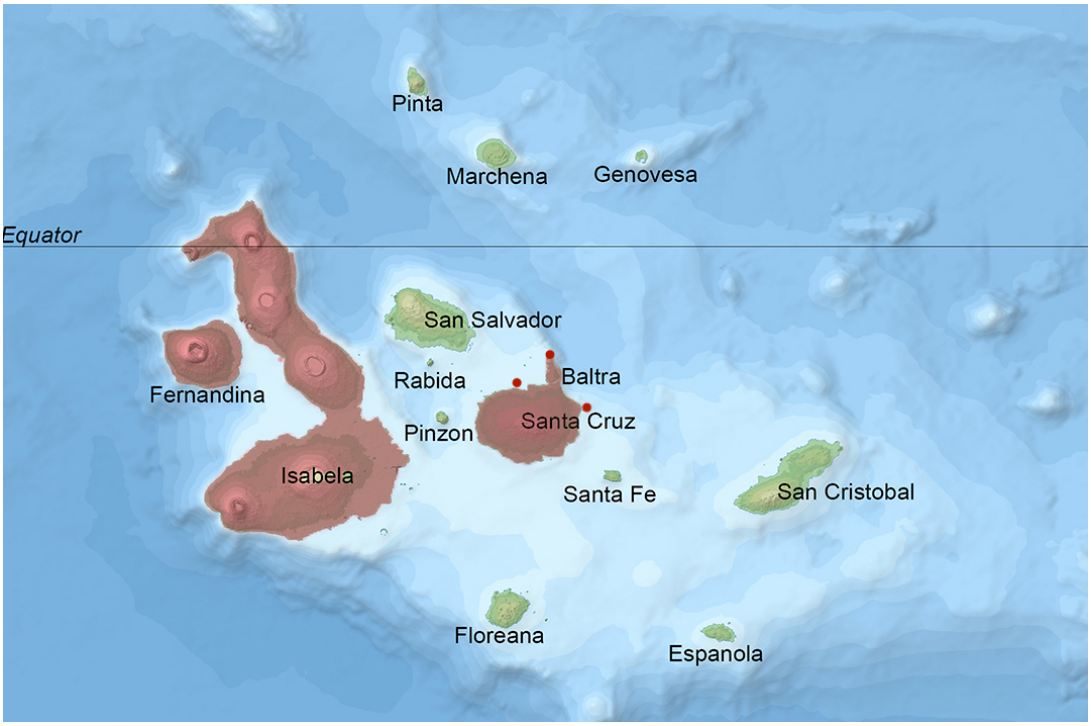
Population (Island)	Hypothesized current population trend
Fernandina	Stable
Isabela	Decreasing
Santa Cruz	Stable or Decreasing
North Seymour	Increasing
Baltra	Stable
South Plaza	Stable

Isabela Island shows decreasing population trend for land iguanas on the table due to the feral cat population on the island. Cat predation has been reported previously on land iguanas in the *Conolophus* genus as well as Marine iguanas (*Amblyrhynchus cristatus*). Feral cats are known to have extremely destructive effects on iguana populations world-wide including the land iguanas of Galapagos islands (Gentile et al., 2016; Mitchell et al., 2002; Nogales et al., 2013). *Conolophus subcristatus* populations on Baltra Island have been reported to have experienced population declines in the past that have been attributed to habitat destruction due to the construction of the military airbase, human disturbance and introduced feral goats, as well as predation from feral cats (Cayot, 2008; Phillips et al., 2005, Woram, 1991). Feral cats have also hypothesized to have had detrimental effects on the Pink iguana (*Conolophus marthae*) population found on Volcan Wolf on Isabela Island. *C. subcristatus* populations coexist with Pink land iguanas on Volcan Wolf, which currently has a living cat population (Gentile et al., 2016), thus it is likely the population of *C. subcristatus* on Volcan Wolf and other parts of the island with cat populations are experiencing declines. Similarly the

population of land iguanas in Cerro Dragon on Santa Cruz Island could be experiencing declines and have been reported to have no juvenile recruitment for the last two decades owing to feral cat predation (A. Llerena, 2018. Pers. Comm).

The population of land iguanas on North Seymour Island appears to be increasing and has been since their introduction to the island in 1932 (Cayot & Menoscal, 1992). However, the rate of increase is likely to have slowed in recent years as the carrying capacity has reached.

Map bellow shows extent of occurrence of land iguanas in the Galapagos archipelago. Island's in red as well as small islands/islets indicated by red dots (Venecia islet, North Seymour Island and South Plaza Island, from left to. right).



Current population trend (tick (✓) one box only)	Increasing	
	Decreasing	
	Stable	
	Unknown	✓

4d. Habitats and Ecology

Provide a summary of the habitats occupied by the taxon, highlighting essential habitats and ecological requirements. It is not necessary to know the details of behavioural traits, etc. unless these are relevant to the taxon's Red List status (e.g., it has a particular life cycle, growth pattern or behaviour that makes it vulnerable to specific threats).

Land iguanas occupy areas that are dry and arid with low growing shrubs and *Opuntia* cactus, as this is their main food source. Iguanas are known to eat at least 30 different plant species (Traveset et al., 2016), thus they are generally found in areas with these species. The iguanas have extremely limited access to fresh water, so acquire most of their liquid intake from the vegetation, primarily the *Opuntia* cactus (Jackson, 1993; Switak, 1998). The arid regions are located on the lower elevations of the islands (Snell & Tracy, 1985). However iguanas are found at high elevations on Fernandina and

Isabela Island. At higher elevations, the vegetation is thicker and the air is more humid. Iguanas on Fernandina in particular are known to migrate approximately 10 kilometres up to the maximum elevation of 1500m. There is plenty of soft soil at the crater of Fernandina, so it is an ideal nesting site for female iguanas and general land iguana habitat. Iguanas are known to descend into the crater of the volcano and lay their eggs inside. It is hypothesized the fumaroles from the volcanic activity help to fertilize the eggs (Werner, 1982, 1983).

Areas with soft soil are generally where land iguanas are found and is their preferred habitat as this is ideal for building their burrows. However, land iguanas have been observed in areas with old broken lava flow with no soft soil available in Fernandina Island, instead they use spaces under rocks and fissures in old lava flow as their resting ground (Carpenter, 1969). They are also opportunistic feeders and occasionally feed on insects and carrion when other preferred food sources have limited availability (Harper, Zabala, & Carrion, 2011).

Land iguanas are diurnal species. They are found basking in the sun not far from their burrows early mornings when the sun has risen. Once their bodies have heated up they are active until the sunset. They retreat back into their burrows overnight. Exact home ranges are unknown. However, male land iguanas have been reported to have larger territories than females (Jackson, 1993).

Elevation in m above sea level	**Upper limit:**	1707	**Depth in m below sea level**	**Upper limit:**	
	Lower limit:	0		**Lower limit:**	

4e. Use and Trade

Provide a summary of any utilization of and/or trade in the taxon (at local, national and international levels). Please remember that the taxon may be utilized or be the focus of local, national or international trade, but if these activities are carried out sustainably they may not actually be a threat to the species; it is therefore useful to record whether this utilization/trade is a likely threat to the global population; this information helps to identify species that are important for human livelihoods, but which may be under threat from factors other than utilization or trade.

If unknown or there is no trade in the taxon, please state this.

Galapagos land iguanas have been attempted to have been smuggled on two occasions. The first occasion was in 2013, a tourist tried to smuggle four land iguanas out of the Galapagos islands (Gentile et al., 2016). The second occasion was in 2015 a man tried to smuggle nine juvenile marine iguanas and two juvenile land iguanas. Both persons had been prosecuted and sentenced (AFP, 2015).

4f. Threats

Provide a summary of the major threats affecting, or likely to affect, the taxon. Try to indicate whether these threats are historic threats that caused past population declines, or current threats affecting the population now, and whether they are likely to affect the population in future.

Please record as much detail about the threats as possible, including the main cause of the threat (the driver), the threat itself, the scale of the threat (e.g., is most of the global population affected, or is the threat affecting only small parts of the population), and the stress this threat places on the taxon (e.g., habitat degradation, loss of breeding sites, loss of prey base, direct mortality, etc.).

Anthropogenic activities since the 1700s have led to major population declines and extirpations of land iguanas in Isabela, Santa Cruz, Baltra and Santiago Island (Fabiani et al., 2011; Tzika et al., 2008).

The most severe threats to land iguanas that have caused previous population declines include the introduction of a number of mammals including feral dogs, cats, rats, pigs, goats and donkeys to many of the islands land iguanas occupy. Feral dogs were the most harmful to land iguanas as they prey on juveniles as well as adult land iguanas (Cayot, 2008). Feral cats are also extremely harmful as they eat land iguana eggs and prey on juvenile land iguanas affecting juvenile recruitment into the population (Phillips et al., 2005). Grazing mammals, such as pigs, goats and donkeys have degraded vegetation on Santiago Island and Baltra Island as well as trampled and destroyed iguana nests (Cayot, 2008; Snell, Snell, & Tracy, 1984). A number of introduced mammals had led to the extinction of land iguanas on Santiago Island. On Isabela and Santa Cruz Island populations of land iguanas had been extirpated or near extirpated in 1975, due to feral dog attacks. Introduced goats, dogs and cats as well as the construction of a military airbase in 1941 on Baltra Island had led to the entire population of land iguanas to wiped out (Cayot, 2008).

Currently there are fewer threats faced by land iguanas. Feral dogs, pigs, goats and donkeys have been greatly reduced and completely eliminated on a number of islands (Cruz et al., 2005, 2009. However, feral cats have proven to be difficult to eradicate from Isabela and Santa Cruz Island which holds the six out of 11 of the land iguana population. Feral cats are the leading threat to land iguanas today (Gentile et al., 2016). They eat land iguana eggs and prey on juvenile land iguanas up to two years of age (Marquez et al., 2010). The resulting lack of juvenile recruitment threatens populations that are exposed to cat predation. Minimal cat control is underway on both islands however this is not enough to remove the cats entirely thus the threat to land iguanas still persists. It is likely land iguanas are currently experiencing population declines and will experience future population declines if the presence of cats still persists. Kumar, Ortiz-Catedral & Llerena (2019) has shown that the Isabela Island population may exhibit an aging population with only 0.89% of the sample size (n= 244) representing juveniles, this is suggested to be the result of juvenile depredation from feral cats.

All islands that land iguanas occupy, apart from Fernandina Island, have introduced mammals. The most abundant introduced mammals are black rats (*Rattus rattus*) and Norway rats (*Rattus norvegicus*) (Clark, 1981; Patton, Yang, & Myers, 1975). Although there are no direct reports of the effect of rats on iguana populations. It is likely rats eat land iguana eggs and they have also been mentioned to prey on juvenile iguanas (Gentile et al., 2016).

Other threats include the possibility of large portions of land iguanas to be wiped out due to the eruption of volcanoes. The Galapagos islands consists of a number of active volcanoes of which the habitat of land iguanas is included. Fernandina and Isabela Island have both experienced past and recent eruptions that have threatened land iguana populations. In fact it has been speculated that some local populations in southern Isabela Island had been exterminated due to a volcanic eruption in the area (Snell, Snell, & Tracy, 1984). Many of the volcanoes in the Galapagos archipelago are still very active and are likely to continuously erupt. The most recent eruption was in June 2018 on Fernandina Island (Vasconez et al., 2018).

4g. Conservation Actions

Provide a summary of the conservation actions currently in place, and **realistic** actions needed to mitigate the major threats to the taxon (if any). This section should not be used to record a full "wish list" of conservation actions for the species; please try to restrict recommendations to those actions that could realistically be implemented and have a good chance of improving the status of the taxon.

Land iguanas are protected under the Appendix II of "Convention on International Trade in Endangered Species of Wild Fauna and Flora" (CITES) (Gabriele Gentile, Ciambotta, & Tapia, 2013).

Historical conservation action:
The first conservation efforts for land iguanas began in 1970s following feral dog attacks on Isabela and Santa Cruz Island. A captive breeding programme was initiated by the Charles Darwin Research Station based in Santa Cruz Island. The few iguanas left in Santa Cruz following the dog attacks were transferred to the breeding station. In 1977 iguanas from the breeding station were translocated to Venecia, an islet off Santa Cruz Island. There also was a breeding program in place to replace the individuals lost in Isabela Island.

In the 1930s land iguanas were translocated from Baltra Island and introduced to North Seymour Island as an experiment (Perkins, 1930). This experimental translocation later benefited the land iguana population on Baltra as land iguanas were driven to extinction here in the 1980s. Soon after, the eradication of invasive mammals in Baltra Island, land iguanas were reintroduced from North Seymour in 1991. All translocations of land iguanas have been successful with founder populations being full established today (Barnett & Rudd, 1983; Cayot & Menoscal 1992; Cayot, 2008).

Mammalian eradication programs have occurred on Baltra (2005), North Seymour (2011), Santiago (1997) and Isabela (1997) Island. (Carrion et al., 2011; Harper, Zabala, & Carrion, 2011; Phillips et al., 2005). Feral cats have been removed from Baltra to protect the land iguana population here (Phillips et al., 2005)). Rats have been attempted to be eradicated on North Seymour Island, however these efforts have been unsuccessful (Harper, Zabala, & Carrion, 2011). The largest mammalian eradication commenced in 1997 and ended in 2006 (Carrion et al., 2011). All grazing animals including goats, pigs and donkeys were eradicated from northern Isabela and the entirety of Santiago Island. The aim of this project was general island restoration for native fauna and flora.

The most recent translocation took place in January 2019. Over 2000 land iguanas were moved from North Seymour Island to Santiago Island in hopes to restore Santiago Island as well as to increase the land iguana population (K. Campbell, pers. Comm, Kumar, Ortiz-Catedral & Llerena, 2019). Before that large translocations of land iguanas have not occurred since the 2000s and the land iguana breeding program was discontinued in 2008 (Cayot, 2008). There are minimal eradication efforts on some islands, however this is directed for general native species of Galapagos rather than specifically for land iguanas (K. Campbell, pers. comm).

5. DATA FOR RED LIST CRITERIA

Record the available data for population sizes, trends, decline rates, ranges, etc. to compare against the IUCN Red List Criteria thresholds.
For full IUCN definitions of "population size", "subpopulation", "mature individuals", "generation length", "reduction", "continuing decline", "extreme fluctuation", "severely fragmented", "extent of occurrence", "area of occupancy", "location", and "quantitative analysis", please refer to the current version of the *Guidelines for Using the IUCN Red List Categories and Criteria*.

5a. Data for criterion A: rate of population reduction

Generation length (please state the unit used).	8.2 years	**Time period used for criterion A** (tick (✓) one box only)	10 years		
			3 generations		
			Time period		

Criteria A1 and A2: % population size reduction over the last 10 yrs or 3 generations:		**Data quality:**	Observed	
			Estimated	
			Inferred	
			Suspected	

Are the causes of this reduction understood? (tick (✓) one box only)	Yes		**Have the causes of the reduction now ceased?** (tick (✓) one box only)	Yes		**Is the reduction reversible?** i.e., is the population now showing signs of recovery? (tick (✓) one box only)	Yes
	No			No			No
	Unknown			Unknown			Unknown

Past population reduction rate based on (select any combination)**:**	Direct observation	
	Index of abundance	
	Decline in area of occupancy, extent of occurrence, and/or habitat quality	
	Actual or potential levels of exploitation	
	Effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites	

Criterion A3: % population size reduction over the next 10 yrs or 3 generations (max. 100 years in future)**:**	✓	**Data quality:**	Projected	
			Suspected	✓

Future population reduction rate based on (select any combination)**:**	Index of abundance	
	Decline in area of occupancy, extent of occurrence, and/or habitat quality	
	Actual or potential levels of exploitation	
	Effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites	✓

Criterion A4: **% population size reduction over the longer time period of 10 yrs or 3 generations, where some time falls in the past and some is projected in to the future** (max. 100 yrs in future)**:**	Unknown*	**Data quality:**	Observed	
			Estimated	
			Inferred	
			Projected	✓
			Suspected	

Population reduction rate based on (select any combination)**:**	Direct observation	
	Index of abundance	
	Decline in area of occupancy, extent of occurrence, and/or habitat quality	✓
	Actual or potential levels of exploitation	
	Effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites	✓

*****It is unknown the percentage of population size reduction for land iguanas. This is mainly due to the fact that there are no formal population estimates for all populations of land iguanas. Especially those on Isabela Island, which is of main concern and holds land iguana populations that are most likely to decline in the next 100 years; this is due to the presence of feral cats. The percentage of population size reduction can only be quantified once population estimates are undertaken for all populations.

5b. Data for criterion B: restricted range

Criterion B1: Extent of occurrence (EOO) in km²:	5255	**Criterion B2: Area of occupancy (AOO)** in km²:	83

Is the population severely fragmented? (tick (✓) one box only)	Yes	✓	If yes, justify this statement in the population text box (refer to habitat fragmentation AND the dispersal abilities of the taxon).	**Number of locations:**	**6**
	No				
	Unknown				

Extent of occurrence	**Continuing decline**		Observed	
			Inferred	
			Projected	
	Extreme fluctuation			

Area of occupancy	**Continuing decline**	✓	Observed	
			Inferred	

	Extreme fluctuation		Projected	✓
Area, extent and/or quality of habitat	**Continuing decline**	✓	Observed Inferred Projected	✓
Number of locations or subpopulations	**Continuing decline** **Extreme fluctuation**		Observed Inferred Projected	
Number of mature individuals	**Continuing decline** **Extreme fluctuation**	✓	Observed Inferred Projected	✓

5c. Data for criterion C: small population size and continuing decline

Population size Number of mature individuals in the global population:				
Is there continuing decline in the population? (tick one box only)	Yes No Unknown		**Rate of continuing decline known?** (tick one box only)	Yes No Unknown
Estimated continuing decline % within 3 years or 1 generation (whichever is the longer time period; max. 100 years in future)**:** **Estimated continuing decline % within 5 years or 2 generation** (whichever is the longer time period; max. 100 years in future)**:** **Estimated continuing decline % within 10 years or 3 generation** (whichever is the longer time period; max. 100 years in future)**:**				

Number of mature individuals in largest subpopulation:		**% of mature individuals in largest subpopulation**	

Extreme fluctuations in number of mature individuals:	Yes	
	No	
	Unknown	

5d. Data for criterion D: small population size or restricted range

Population size Number of mature individuals in the global population:					
Area of occupancy (AOO) in km²:		**Number of locations** :		**Is there a plausible threat that could rapidly push the taxon towards extinction?**	Yes / No / Unknown

5e. Data for criterion E: quantitative analysis

Has a quantitative analysis predicting probability of extinction been carried out? (e.g. Population Viability Analysis)	Yes	
	No	✓
	Unknown	

Probability (%) of extinction within the next 10 years or 3 generations (use the longer time period; max. 100 years in future)	
Probability (%) of extinction within the next 20 years or 5 generations (use the longer time period; max. 100 years in future)	
Probability (%) of extinction within the next 100 years	

6. RED LIST ASSESSMENT

Assess the taxon using the information and data recorded in section 4 and 5, and following the *IUCN Red List Categories and Criteria: version 3.1.* and current version of the *Guidelines for Using the IUCN Red List Categories and Criteria* for guidance on applying the IUCN criteria.

6a. Red List Category & Criteria

Tick (✓) one of the following Red List categories, For taxa qualifying for a threatened category (CR, EN or VU), record all criteria and subcriteria met. For the NT category, record all criteria and subcriteria nearly met:

	Extinct (EX)	Date last seen in wild (day/month/year)	
	Extinct in the Wild (EW)	Date last seen in wild (day/month/year)	
	Critically Endangered (CR)	Criteria met for CR	
	Endangered (EN)	Criteria met for EN	
✓	**Vulnerable (VU)**	Criteria met for VU	A3,B1
	Near Threatened (NT)	Criteria nearly met for NT	
	Least Concern (LC)		
	Data Deficient (DD)		
	Not Evaluated (NE)		

Is this taxon Possibly Extinct? (applies to CR taxa only)	Yes	
	No	✓
	Unknown	

6b. Rationale for the assessment

Provide a summary of the reasons why the taxon qualifies for the category and criteria recorded in section 6a. Include any population or range information used, inferences, assumptions, etc. For NT specify what criteria were nearly met and for DD state what little information is known. Please refer to the *IUCN Red List Categories and Criteria: version 3.1.* and the current version of the *Guidelines for Using the IUCN Red List Categories and Criteria* for guidance on definitions of terms and applying the IUCN criteria).

Land iguanas meet the threat category of Vulnerable under criteria A3. Given the insular nature of land iguanas being restricted to six islands of the Galapagos archipelago, it would generally be considered as a high risk extinction species. This is difficult to quantify given the unpredictability of stochastic events on islands. The main threat however is the effect of introduced predators namely cats on populations of land iguanas. There are 11 confirmed populations of land iguanas, 6 of these are threatened with cats. These certain populations are thus subject to reduction in juvenile recruitment, resulting in potential for projected population declines and also currently suspected population declines. It has been documented previously that heavy predation and lack of juvenile recruitment can wipe out entire populations of any iguana species including Galapagos land iguanas (Gentile et al., 2016; Mitchell et al., 2002; Nogales et al., 2013). Further, land iguanas on Cerro Dragon on Santa Cruz Island have had no records of successful breeding or juvenile recruitment in the last 20 years (A. Llerena, 2018. Pers. Comm). It is possible that this could be the case for other populations of land iguanas with the presence of cats.

Land iguanas meet the threat category of Vulnerable under criteria B1 as well. If the extent of occurrence of the species is estimated to be less than 20,000 km^2 and shows at least 2 characteristics of: a.) severe fragmentation (exists <10 locations), b.) continuing decline whether observed or projected of any one of: the extent of occurrence, area of occupancy, area extent, quality of habitat, number of locations or subpopulations and number of mature individuals, c.) extreme fluctuations in any of: the extent of occurrence, area of occupancy, number of locations or subpopulations or number of mature individuals, then the species would be classified as Vulnerable. This is a likely case for the land iguana. For all populations of land iguanas, I have included the entire size of the island in accordance to the definition of “extent of occurrence” under section III. DEFINITIONS of the Guidelines for Using the IUCN Red List Categories and Criteria. However, barring Venecia and Cerro Dragon from Santa Cruz Island due to the fact that they are very small (<1 km^2) isolated populations. The total extent of occurrence for land iguanas is approximately 5255 km^2. This figure falls well under 20,000 km^2 so can potentially fall under the Vulnerable category as long as two of the characteristics I have mentioned previously apply. Something worth noting is the figure of 5255 km^2 being very close to 5000 km^2, as if the extent of occurrence falls under 5000 km^2 and the species occurs no more than five locations, then it would fall under the Endangered category. There are six localities of land iguanas being the islands they occupy. Therefore, land iguanas would fall under Vulnerable for B1. Continuing decline observed, inferred or projected, in extent of occurrence and quality of habitat is likely to occur in the next few decades given the current circumstances of iguana populations that are exposed to feral cats. Land iguana habitats on Isabela Island and Cerro Dragon on Santa Cruz Island can be deemed as poor quality due to the presence of cats. If feral cats persist it is likely populations can reduce in size and some populations have the potential to be extirpated if cat control is not in place.

Assessment Date:	DD	MM	YYYY
	31	01	2019

Assessors' Names:	Given Name(s)	Family Name	Email Address	Institution
	Kirtana	Kumar	kirtana.k.9@gmail.com	Institute of Natural and Mathematical Sciences

6c. Changes in Red List status

Check the IUCN Red List web site (www.iucnredlist.org) to find out whether the taxon has previously been assessed.

Has this taxon been assessed for a previous IUCN Red List?	Yes	✓	**If yes, what was the previous assessment?**	*Conolophus subcristatus* 1996
	No			
	Unknown			
If yes, has the taxon changed category since its last assessment?	Yes		**If no, have the criteria changed?**	Yes
	No	✓		No ✓

Reason for change in category:

Genuine change	Recent change		**Non-genuine change**	New/better information available ✓
	Change since first assessment			Taxonomic change
				Incorrect application of criteria previously
				Criteria thresholds changed since previous assessment

7. LITERATURE REFERENCES

Provide a list of all published and unpublished reference sources used for the information recorded above. Please provide full references, and try to avoid abbreviations (e.g., write *Conservation Biology* rather than *Cons. Biol.*).

AFP. (2015). Mexican arrested for trying to smuggle iguanas from Galapagos Islands. *The Telegraph*.

Barnett, D. B., & Rudd, L. R. (1983). Feral dogs of the Galapagos islands: Impact and control. *Animal Studies Repository*, 4.

Carrion, V., Donlan, C. J., Campbell, J. K., Lavoie, C., & Cruz, F. (2011). Archipelago-wide island restoration in the Galapagos islands: reducing costs of invasive mammal eradication programs and reinvasion risk. *Plos one*, 6.

Cayot, J. L. (1992). Land iguanas return to Baltra. *Noticias De Galapagos*, 11-13.

Cayot, J. L. (2008). The restoration of giant tortoise and land iguana populations in Galapagos. *Journal of Science and Conservation in the Galapagos Islands* (65), 39-43.

Clark, D. A. (1981). Foraging patterns of black rats across a desert-montane forest gradient in the Galapagos Islands. *Biotropica*, 182-194.

Cruz, F., Carrion, V., Campbell, K. J., Lavoie, C., & Donlan, C. J. (2009). Bio-economics of large-scale eradication of feral goats from Santiago island, Galápagos. *The Journal of Wildlife Management*(2), 191.

Cruz, F., Donlan, C. J., Campbell, K., & Carrion, V. (2005). Conservation action in the Galapagos: feral pig (*Sus scrofa*) eradication from Santiago Island. *Biological Conservation*, 121(3), 473-478.

Duméril, C., Bibron, G., & Duméril, A. H. A. (1839). *Erpétologie générale: ou, Histoire naturelle complète des reptiles* (Vol. 5): Roret.

Fabiani, A., Trucchi, E., Rosa, S., Marquez, C., Snell, H. L., Snell, H. M., Gentile, G. (2011). Conservation of Galapagos land iguanas: genetic monitoring and predictions of a long-term program on the island of Santa Cruz. *Animal Conservation*, 14(4), 419-429.

Fitzinger, L.-J. (1843). *Systema reptilium* (Vol. 1): Braumüller & Seidel.

Gentile, G., Ciambotta, M., & Tapia, W. (2013). llegal wildlife trade in Galapagos: molecular tools help the taxonomic identification of confiscated iguanas and guide their rapid repatriation. *Conservation Genetics Resources*, 5(3), 867-872.

Gentile, G., Marquez, C., Snell, H. L., Tapia, W., & Izurieta, A. (2016). Conservation of a New Flagship Species: The Galápagos Pink Land Iguana. *Problematic Wildlife*. 315-336

Gentile, G., & Snell, H. (2009). *Conolophus marthae sp.*nov. (Squamata, Iguanidae), a new species of land iguana from the Galápagos archipelago. *Zootaxa* (2201), 1-10.

Gray, J. E. (1831). Description of a new species of Amblyrhynchus of Mr. Bell, in the British Museum. *Zoological Miscellany*, 10(6).

Gray, J. E. (1845). Catalogue of the Specimens of Lizards in the Collection of the British Museum: order of the Trustees.

Harper, G. A., Zabala, J., & Carrion, V. (2011). Monitoring of a population of Galápagos land iguanas (*Conolophus subcristatus*) during a rat eradication using brodifacoum. Island Invasives: *Eradication and Management*, 309-312.

Heller, E. (1903). Reptiles (Vol. 14): The Academy.

Jackson, M. H. (1993). *Galapagos, a natural history*. University of Calgary press. Kumar, K., Ortiz-Catedral, L., & Llerena, A. J. (2019). Conservation status and demographics of the Galapagos Land Iguana (*Conolophus subcristatus*) (Masters of Natural Science in Conservation Biology), Massey Univeristy, Auckland, New Zealand.

Marquez, C. M., Munoz, E. A., Gentile, G., Tapia, W. H., Zabala, J. F., Naranjo, S. A., & Llerena, A. J. (2010). Estado poblacional de las iguanas terrestres (*Conolophus subcristatus, C. pallidus y C. marthae: Squamata, Iguanidae*), Islas Galápagos. *Bioletín Técnico: Serie Zoológica*, 9(6), 25-43.

Mitchell, N., Haeffner, R., Veer, V., Fulford-Gardener, M., Clerveaux, W., Veitch, C. R., & Mitchell, G. (2002). Cat eradication and the restoration of endangered iguanas (*Cyclura carinata*) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group., 206-212.

Nogales, M., Vidal, E., Medina, F. M., Bonnaud, E., Tershy, B., Campbell, K. J., & Zavaleta, E. S. (2013). Feral Cats and Biodiversity Conservation: The Urgent Prioritization of Island Management. *BioScience*, 63(10), 804-810.

Patton, J. L., Yang, S. Y., & Myers, P. (1975). Genetic and morphologic divergence among introduced rat populations (*Rattus rattus*) of the Galapagos Archipelago, Ecuador. *Systematic Biology*, 24(3), 296-310.

Perkins, C. B. 1930. Galapagos Islands Trip December 4th 1931 to February 27th 1932. Archive PER 5924.63 Charles Darwin Research Station Library, Puerto Ayora, Santa Cruz, Galapagos Islands, Ecuador.

Phillips, R. B., Cooke, B. D., Campbell, K., Carrion, V., Marquez, C., & Snell, H. L. (2005). Eradicating feral cats to protect Galapagos land iguanas: methods and strategies. *Pacific Conservation Biology*, 11, 257-267.

Snell, H. H., Snell, M. H., & Tracy, C. R. (1984). Variation among populations of Galapagos land iguanas (*Conolophus*): contrasts of phylogeny and ecology of phylogeny and ecology. *Biological Journal of the Linnean Sociey* (21), 185-207.

Steindachner, F. (1876). Die Schlangen und Eidechsen der Galapagos-Inseln: Wien :K.K. *Zoologisch-botanischen gesellschaft.*

Tzika, A. C., Rosa, S. F. P., Fabiani, A., Snell, H. L., Snell, H. M., Marquez, C., . . . Milinkovitch, M. C. (2008). Population genetics of Galapagos land iguana (*genus Conolophus*) remnant populations. *Molecular Ecology*, 17(23), 4943-4952.

Vasconez, F. J., Ramón, P., Hernandez, S., Hidalgo, S., Bernard, B., Ruiz, M., . . . Ruiz, G. (2018). The different characteristics of the recent eruptions of Fernandina and Sierra Negra volcanoes (Galápagos, Ecuador). *Volcanica*, 1(2), 127-133.

Werner, D. I. (1982). Social organization and ecology of land iguanas, *Conolophus subcristatus*, on isla Fernandina, Galapagos. Park Ridge, USA: Neyes Publications.

Werner, D. I. (1983). Reproduction in the iguana *Conolophus subcristatus* on Fernandina island, Galapagos: clutch size and migration costs. *The American Naturalist*, 757-775.

Woram, J.M. 1991. Who killed the iguanas? *Noticias de Galapagos*. 50, 12-17

DISCUSSION

Having an up-to-date species assessment on the IUCN Red List is imperative for informing conservation decisions as well as informing legislative frameworks to address conservation of the given species (Hoffmann et al., 2008; Vié et al., 2009). Consequently, having a well-informed Red List assessment is likely to decrease the extinction risk of the species and increase its ability for recovery.

Galapagos land iguanas have been assigned the threat category of 'Vulnerable' since 1996. This means the evidence available indicates that land iguanas meet at least one of the Criterion A to E, these criteria are provided in the 2001 IUCN Red List Categories and Criteria version 3.1 and also the subheadings in the questionnaire. At current, there are five sub headings (A to E) under each threat category and at least one must apply to the species to be eligible for a category of threat. The Red List status is determined by factors related to reduction in population size, current geographic range of species, a population estimated to be fewer than 10, 000 mature individuals, a population very small or restricted and a quantitative analysis showing the probability of extinction in the wild to be at least 10% within 100 years (IUCN, 2017). Although given the threat category of Vulnerable (1996), there is no information or evidence to support this. The information given on IUCN Red List website has remained the same since 1996. Crucial information of the population size and trends, geographic range, threats and conservation actions to inform this given category of 'Vulnerable' is lacking.

Information and research on Galapagos land iguanas have accumulated over the last few decades. There is now a significant amount of valuable information to re-evaluate the land iguanas threat category. I have used this information as a basis to re-evaluate and update the current Red List entry for Galapagos land iguanas. I have concluded that the threat category of 'Vulnerable' is still applicable to land iguanas today. I have provided the information in the questionnaire as evidence to support this category.

Specifically Criterion A and B are met for land iguanas. Criterion A is based on the rate of population reduction whether observed, estimated, inferred, suspected or projected. For land iguanas I have suggested the likely case of projected population reduction in the near future, this is under the assumption that the presence and predation of feral cats persist as it is currently, this is likely, as stated earlier the eradication of cats on Isabela Island is not a realistic outcome owing to its large size (Gentile et al., 2016). Criterion B is based on restricted range of the species. I have calculated the extent of

occurrence of land iguanas to be approximately 5255 km^2, this is the total area of the islands they occupy. I have also calculated the area of occupancy, which is where land iguanas specifically occur on the islands they occupy to equal to approximately 83 km^2. This calculation was aided by (Marquez et al., 2010). For land iguanas to receive the threat category of Vulnerable under Criterion B, their extent of occurrence must be under 20,000 km^2. This is therefore applicable. As for the area of occupancy, land iguanas must occupy an area under 2,000 km^2. This is also applicable. Additionally, the Vulnerable category can only be granted to land iguanas under criterion B if they also show two of either a.) severely fragmented population/ or b.) continuing decline observed, estimated, inferred or projected in any of : extent of occurrence, area of occupancy, area extent or quality of habitat, number of locations or subpopulations or number of mature individuals or c.) extreme fluctuations in any of: extent of occurrence, area of occupancy, area extent or quality of habitat, number of locations or subpopulations or number of mature individuals. Both b.) and c.) are applicable for land iguanas, it is likely based on the information I have provided that land iguanas are continuing decline observed, inferred or projected, in extent of occurrence, quality of habitat, number of sub populations and mature individuals is likely to occur in the next few decades given the current circumstances of some iguana populations particularly those subject to cat predation. To conclude my evaluation and this chapter, the given category of threat for land iguanas is Vulnerable under Criterion A3 and B1.

At the time this thesis was submitted, the release of land iguanas onto Santiago Island had occurred (January, 2019). The land iguanas were collected from North Seymour Island and then kept in quarantine on Santa Cruz Island prior to their release. Link for press release on land iguana translocation 2019: https://edition.cnn.com/travel/article/galapagos-island-iguanas-scli-intl/index.html. See also following page for sample of a press release published on the Guardian.



Figure 4.2: Photographs of land iguanas after their release on Santiago Island. Top left shows female land iguana near *Opuntia* cactus. Top right shows land iguana tracks at Bucanero coast. Bottom shows male land iguana under *Opuntia* cactus

CHAPTER 5

Pilot Studies of Land Iguanas and Conclusions and Recommendations



Figure 5.1: Photograph of a Galapagos hawk (*Buteo galapagoensis*) on a Galapagos land iguana on Fernandina Island. Photo: K. Kumar

ABSTRACT

Galapagos land iguanas have been subject to anthropogenic disturbances since the 17th century. Off which many have been minimized, however feral cat populations are the leading threat to Galapagos land iguanas today. 55% of the population are currently exposed to cat predation, consequently reducing the juvenile recruitment of these populations which can result in projected population declines. There has been a lack of research that investigates the effect of introduced predators on land iguana populations. More so there is limited information given in their IUCN Red List Assessment, an issue than can prevent effective management of the species. The IUCN SSC Iguana Specialist Group and the Galapagos National Park have been discussing the translocation of land iguanas form North Seymour to Santiago Island since 2014, for the purpose of restoring Santiago Island and restocking the population of land iguanas. The research gaps needed to aid and inform the translocation are addressed in this thesis. This chapter briefly summarizes the findings of my thesis and presents two pilot studies. Finally to conclude this thesis, a set of recommendations and research priorities are given for ensuring a successful translocation as well as for future land iguana studies in the Galapagos islands.

INTRODUCTION

This chapter discusses some components that were investigated in the field, off which are connected to the reintroduction of land iguanas to Santiago Island. However this was explored over a short time due to time and funding constraints prior to completing this thesis. The information provided in this thesis has helped Island Conservation and the Galapagos National Park to plan and implement the reintroduction of land iguanas to Santiago Island, see Figure 4.2. The first release of land iguanas occurred on January 4th 2019. Although at this time I did not have the opportunity or resources to participate in the translocation, I present priority areas of research that will contribute to further translocations to Santiago Island and the conservation of the species as a whole. The documents detailing components of my research in the thesis if provided in the Appendix. This includes field reports on our population estimates and sampling on North Seymour Island. These reports give earlier estimates that were based on preliminary models.

The findings in Chapter 2 allow us to make some conclusions on the status of land iguana populations. Although our analysis did not quantify population trends, our findings allow us to infer current and future trends which are also covered in Chapter 4. The most important finding is how severely the characteristics of land iguana populations differ between cat-present and cat-absent populations (Isabela and Cerro Dragon). Cat-present populations exhibit an aging population with only 1% of the population representation juveniles when compared with 12% of juveniles in cat-absent populations, this is also an indicator of a declining population trend. Similarly cat-present populations have significantly larger SVL's and body condition indices further indicating heavy juvenile predation and low juvenile recruitment in cat-present populations as only large and fit individuals are left. The results of this analysis contributes to our predicted population trends and population projections covered in Chapter 4.

Chapter 3 was aimed at providing the information necessary to aid the planned translocation of land iguanas from North Seymour Island to Santiago Island. Additionally I investigated the use of Mark-recapture and distance surveys for estimating population size of land iguanas. Our findings conclude using Mark-recapture is intensive and costly and should and therefore should only be used for populations that are in smaller islands with moderate to low densities. Distance sampling provided reliable estimates to inform the Galapagos National Park on the current numbers of iguanas on North

Seymour Island. Distance surveys are recommending due to the relatively low level of investment in terms of cost and effort.

Current population estimates, population status and characteristics were provide as a basis for appropriately choosing individuals for the translocation. Attaining a successful translocating involves choosing individuals that are well suited and robust to withstand the translocation (Letty, Marchandeau, & Aubineau, 2007). Hence characteristics such as large SVL, and optimal body condition indices are some important determinants of individuals that are more likely to survive the translocation process. Additionally using Fernandina Island as a model population allows us to uncover any unfavourable population characteristics. For instance, I found that the land iguana population from North Seymour Island represented a very dense population on a small island, this is assumed to have contributed to the significantly lower body condition indices on this island when compared to individuals from Fernandina Island of the same length. Additionally, the high juvenile percentage of 28.65% is concerning as this indicates very recent population growth that the island may not be able to withstand in the near future. Our findings show herbivore pressure from land iguanas on the micro-endemic species of cacti *Opuntia echios zacana,* which are especially vulnerable to extinction due to their inability to protect themselves having evolved without grazing mammals (Biggs, 1990), this in turn can threaten the balance of the ecosystem. Finally, the findings in this thesis support the need for the translocation of land iguanas from North Seymour Island to relieve the pressure of the island whilst safe guarding the land iguana population and aiding the restorative process on Santiago Island.

ADDITIONAL PILOT STUDIES CONDUCTED DURING THE DATA COLLECTION PERIOD

Burrow density surveys on Fernandina and North Seymour:

Indices of population size and animal abundance can be achieved by surveying the individuals in a population directly or in some cases indirect survey methods can be used. For instance, counting the burrows of crustaceans has been used to form an index for the abundance and density of the population of crustaceans (Schlacher et al., 2016). This form of estimating animal abundance is less invasive and reduces possible habitat destruction and stress encountered by the animal. However, burrow counts as well as assessing the occupancy of the burrows is required to acquire reliable

estimates. For example, signs of fresh tracks, scats and cleared vegetation around the burrow indicate an active or occupied burrow for semi-fossorial animals (Hubbs, Karels, & Boonstra, 2000) such as Townsend's ground squirrels (*Spermophilus townsendii*) (Van Horne et al., 1997) and European badgers (*Meles meles*) (Gese, 2001).

Indirect density estimates as well as estimating morphological measurements have also been obtained for Cuban iguanas (*Cyclura nubila nubile*) by counting burrows (Daniel & Berovides, 2004). Daniel & Berovides (2004) findings suggest that the amount of burrows greatly affects the population size and has mentioned that the availability of refuges such as burrows or tree holes can regulate populations of certain species of birds and fish. This suggest that burrow occupancy and burrow density could offer valuable insights on population trends of burrowing species.

We performed a pilot study on burrow occupancy of land iguanas from North Seymour and Fernandina Island. We hypothesized that the population of land iguanas on North Seymour Island is near carrying capacity and is possibly unfavourable dense. By collecting information of the occupancy of burrows this gives insights on the habitat usage of land iguanas from the different populations, this in turn can tell us if there are limited refuges for land iguanas or if there are substantial refuges to support the population. Additionally collecting burrow data of land iguanas can allow for understanding the relationship between land iguana density and burrow density.

Firstly upon visual inspection both North Seymour and Fernandina Island appeared to have numerous burrows where iguanas occur, they were not hard to come by. There also appeared to be an increased number of burrows in close proximity around the crater rim of Fernandina Island and on Platforma on North Seymour Island.



Figure 5.2: Photographs showing examples of land iguana burrows. Top photo shows several iguana burrows on Fernandina Island. Bottom photo shows an occupied burrow at the top near a vacant possibly damaged burrow at the bottom. Photos: L. Ortiz-Catedral

Burrows of land iguanas were surveyed by walking along an unspecified length of the land and noted if they were either “occupied” or “vacant”. To determine if the burrow was “occupied”, a combination of visual observation for the iguana, odour, checking for fresh tracks and scats gave an indication that the burrow was in use. Burrows with cobwebs over the entrance and that lacked odour or appeared to be damaged/ broken in and empty were deemed as “vacant” and out of use. A total of 998 burrows were observed and investigated for occupancy on Fernandina Island. 418 burrows were observed and investigated on North Seymour Island. 12% of burrows on Fernandina Island were occupied, meaning they were active and in use by the iguana currently or very recently (within 1-2 days). 28% of the

occupied burrows were found with an iguana in it. On North Seymour Island 38% of the burrows sampled were classified as occupied and 17% of the occupied burrows were occupied by an iguana at the time. A Chi-Squared reveals significant differences in burrow occupancy between Fernandina and North Seymour Island (χ^2 = 118, *df* = 1, P = <0.0001).

Table 5. 1: Summary of burrow occupancy on North Seymour and Fernandina Island.

	North Seymour	Fernandina
Percentage of occupied* burrows	38%	12%
Percentage with iguana inside	17%	28%
Number of burrows surveyed	418	998

*Occupied meaning a burrow that is in recent use by an iguana, but without an iguana in it



Figure 5.3: Satellite image showing land iguana burrow locations. Top shows burrow locations (in white) on North Seymour Island and below shows burrow locations on Fernandina Island.

Overall vacant burrows numbered high for both populations (over 50%). A high number of vacant burrows could mean that land iguanas often dig new burrows and abandon their previously used burrows. It is unknown whether land iguanas use the same burrows every year or if they dig new ones, it is also unknown whether iguanas use other individuals burrows. However, we did encounter when

capturing iguanas during our Mark-recapture surveys that many would retreat into a burrow nearest to them and were often startled by another iguana occupying the burrow, at times the iguana fleeing would have to leave the burrow as it was an unwanted guest. It is known land iguanas are territorial especially males thus this along with our observations suggest land iguanas don't use other individuals burrow. There are records of black iguanas (*Ctenosaura similis*) in Costa Rica using burrows for consecutive years on end, it was also found that black iguanas retreated back to the same burrow and never a new burrow suggesting they are territorial with their burrows (Burger & Gochfeld, 1991).

Another possibility for the large amount of vacant burrows could be that the vacant ones are old and damaged burrows that are not usable, it is possible the soil or the ground has caved on the inside. Lastly the development of eggs of land iguanas is influenced by the water content in the soil they are laid in (Snell & Tracy, 1985). Thus the soil water potential is an important factor in the laying site of land iguanas. The vacant burrows could be vacant because they are old burrows that have now reduced water potential and thus are no longer suitable laying sites for land iguanas.

Our expectations of having higher burrow occupancy in the North Seymour population of land iguanas was met. This suggests that land iguanas on North Seymour Island are more constrained in their habitat and occur densely, whereas we see land iguanas from Fernandina Island occupying much fewer burrows suggesting there are no space constraints on this population and iguanas choose to not nest in close proximity to other iguanas. Lastly finding a high density of burrows on Platforma, North Seymour perhaps gives evidence that land iguanas prefer using burrows over crevices and rocks, as Platforma on North Seymour Island is the only area on the island that contains soft soil which is ideal for building burrows, the rest of the island is largely rocky. Hence there is a high density of burrows on Platforma as it is only a small section of North Seymour Island with soft soil to support a large population of land iguanas. Similarly burrow density is particularly high around the crater of Fernandina as the soil is especially soft and optimal for burrow building, which explains the especially high density of land iguanas at the crater when compared to other areas on Fernandina Island. One can thus expect that finding a high density of burrows is correlated with a high density of iguanas in the same area. Although this is not quantified as such in the field, this can be explored in future research.

Burrow occupancy studies on iguanas are limited (Burger & Gochfeld, 1991; Daniel & Berovides, 2004; Sementelli et al., 2008). The purpose of performing burrow surveys on North Seymour

and Fernandina Island was to see if the land iguana population on North Seymour Island was in fact too dense. By comparing the burrow occupancy between the two islands, it allows us to gain insight on the generally burrow occupancy of land iguanas. The burrow occupancy results from Fernandina provide an index of what land iguanas prefer in the most ideal and natural environment. Burrow density data can provide researchers with information relating to social organization and habitat use of land iguanas, which are severely understudied aspects of Galapagos land iguanas.

La Bomba (Santiago) habitat survey

It is recommended by the IUCN/SSC Reintroduction Guidelines to survey and assess the suitability of the habitat in the proposed destination area prior to a conservation translocation (IUCN/SSC, 2013). Habitat suitability is crucial for successful establishment of a translocated population. 16% to 50% of failed translocations are a consequence of poor habitat quality (Baling et al., 2016). Consequently failure to assess the release site for suitability can result in a failed translocation. For instance, translocations of Green and Golden Bell Frogs (*Litoria aurea*) in the Greater Sydney area have been unsuccessful due to a number of reasons that are associated to poor habitat quality in the release site. A combination of predation, disease and inadequate over-winter habitat had contributed to failed translocations (White & Pyke, 2008). Whereby if these habitats were assessed for suitability sufficiently the outcome of these translocations may have been successful.

Additionally, assessing the availability of food sources for the proposed translocated animal is sometimes necessary for food sources that may limit the growth and long-term survival of a translocated population, especially if the release site is an island as dispersal abilities are limited for many terrestrial species. Additionally, assessing food sources and availability will provide information on the number of translocated individuals an island can support. For example (Cheyne, 2006) performed habitat surveys involving fruit abundance, fruit productivity and tree density prior to translocations of Agile gibbons (*Hylobates agilis albibarbis*) to Mintin island in Indonesia. Based on the habitat quality assessment, the study concluded the release site (Mintin island) could support a maximum of 19 agile gibbons.

Studies such as these can be applied to other species with the same circumstances like the land iguana. The reintroduction of land iguanas to Santiago Island could be considered as a controversial

operation. The reasons being that land iguanas have not occurred here for around 200 years, there is also not genetic information of the land iguanas that used to inhabit Santiago Island. The vegetation on the island has also been degraded since the introduction of grazing mammals in 1800's (Cruz et al., 2009; Cruz et al., 2005), thus the habitat that the land iguanas used to occupy may be significantly different today. Although, land iguanas have proven to be highly adaptable and with their opportunistic nature during past translocations (Cayot, 2008; Tzika et al., 2008) it is recommended to perform habitat suitability surveys on Santiago Island to reassure the success of future translocations.

One of the main release sites for the proposed translocation of land iguanas from North Seymour Island is La Bomba (0°11'10"S, 90°41'59"W), decided by the Galapagos National Park. La Bomba is situated on the northern-central coast of Santiago Island, the vegetation is dominated by the endemic Galapagos Prickly Pear cactus (*Opuntia galapageia*), native Palo Santo tree (*Bursera graveolens*) and a endemic Galapagos shrub species *Croton scouleri* (Helsen et al., 2009; Patton, Pardiñas, & D'Elía, 2015). Both *Opuntia galapageia* and *Bursera graveolens* are part of the Galapagos land iguana diet (Traveset, 1990).

I conducted a pilot study to assess the density of iguana edible *Opuntia* cactus (*Opuntia galapageia var. galapageia*) as this is one of the main food sources for land iguanas in the islands they inhabit (Christian, Tracy, & Porter, 1984; Costantini et al., 2009), see Figure 5.4. We recorded cacti that were only at least three metres in length as young cacti below three metres high are not suitable as a food source for land iguanas due to their especially sharp and protective spines at early life stages (Biggs, 1990). The purpose of this was to assess if whether there are sufficient cacti available as a food source for the founder land iguana population.



Figure 5 4: Multiple sizes of *Opuntia galapageia* on La Bomba, Santiago Island. Notice top image shows cacti with flower buds on the right side. Photos: K.Kumar

The study was conducted on the 19th of January 2018. It was a point transect survey with each point at least 100 metres apart. The radius of each point was measured at 25 m, thus the area we surveyed for each point is 1963.50 m^2. A total of 10 points were surveyed, so the total area surveyed for the study is equalled to 19,645 m^2 or 1.96 hectares. Two size categories of cacti were recorded. One was 3m and the other was 3-7 m, the study was restricted to these sizes as these cacti are of flowering age and their spines are much smaller which allows iguanas to be able to chew through them. However, there were several cacti below 3 m based on our observations in the field.



Figure 5.5: Photographs of young *Opuntia galapageia* on Santiago Island. Photos: K. Kumar

Our sample size consists of 311 cacti in total, 224 were 3 m cacti and 87 were 3-7 m cacti. With our results I estimated a density of 31 large cacti per area surveyed (1963.5m^2). This equates to 1 large cactus per 63.14 m^2. 72.03% of these were 3 meters and the remaining percentage was 3 to 7 metres. Additionally there were numerous smaller cacti below 3 metres at a range of heights. These results indicate a significant amount of regenerative growth at varying times. I speculate that the cacti at 3-7 m (27.97%) represents cacti that survived during the period grazing mammals were present prior to their eradication. As for the cacti at 3m (72.03%) this represents growth in that last 13 years since the eradication of grazing mammals in 2006 (Carrion et al., 2011). Hence, there is a much higher occurrence of these with the absence of grazing pressure from introduced mammals.

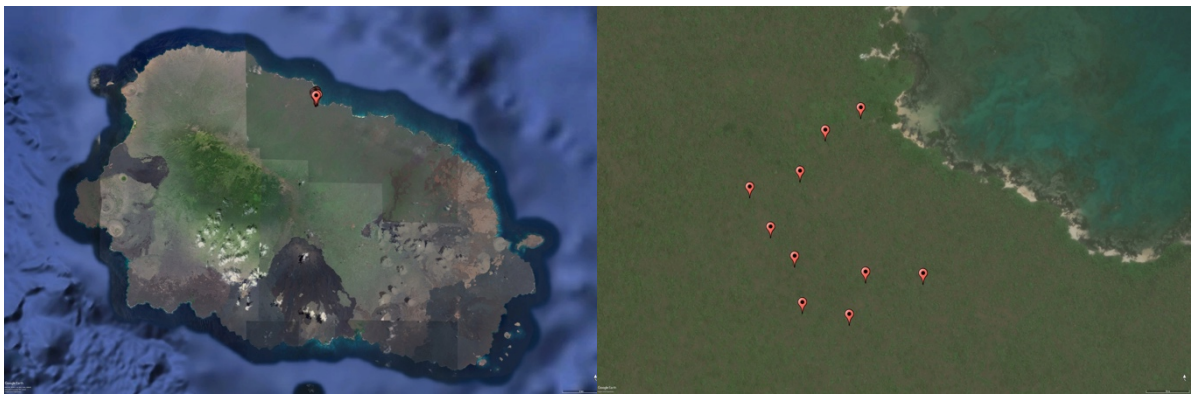


Figure 5.6: Habitat survey site, La Bomba on Santiago Island

These results suggest there is a substantial improvement of cactus regeneration since the eradication of introduced mammals. This indicates a sufficient supply of food for the founder population of land iguanas with sign of available food sources even in the next 10 years as many of the cacti are under 30 cm in length that are currently inedible, however will provide as a food source for iguanas once they are of flowering age and taller, which takes two to three years to reach approximately three metres (Biggs, 1990).

Post-translocation monitoring is encouraged to ensure the long-term persistence of land iguanas on Santiago Island. Additionally, if certain sites are lacking preferable features for land iguanas such as soft soil for building nests, these can be artificially added to assist their introduction. For instance prior to the translocation of Green and golden bell frogs (*Litoria* aurea) to Botany in the Greater Sydney area, artificial ponds and other habitats were built in order to assist the establishment of the founder bell frog population (White & Pyke, 2008).

FUTURE RECOMMENDATIONS

Based on previous studies of land iguanas and the results of our research presented in this thesis, I have provided a set of recommendations concerning the conservation and management of Galapagos land iguanas. The following suggestions are currently lacking in research and aim to provide the means for successfully managing this species across the Galapagos archipelago.

Study on effects of feral cats on populations of land iguanas

At the onset of this study, experimental cat control was the main objective. This was aimed at Isabela Island and Cerro dragon on Santa Cruz Island. However, logistically this was not a feasible

project. Consequently, the objectives were instead altered to inform the translocation of land iguanas as this was planned to commence in the near future as planned by the Galapagos National Park. Although this thesis does look into the effects of cats populations on land iguana populations in Galapagos, it is not a direct study. Data on cat populations in the Galapagos archipelago is lacking or outdated. Furthermore the direct effect of cat predation on land iguanas has never been recorded or measured despite feral cats being one of the most adverse threats to iguanas worldwide (Alberts, Carter, & Hayes, 2004; Iverson, 1978; Mitchell et al., 2002; Nogales et al., 2013).

Of most concern is the Isabela Island population. Isabela is the largest island and holds over 50% of the Galapagos land iguana population. Feral cats are known to inhabit Isabela Island since the 1800's. (Konecny, 1987) reports one of the few population studies of feral cats on Galapagos islands on one site in Isabela Island (Tagus Cove) and Santa Cruz Island (Cerro Colorado). This study provides information on the home range and activity patterns of cats from the two sites mentioned earlier. This study is small scale as it only covers small portion of Isabela Island and Santa Cruz Island, it is also outdated and does not contain information on the density and range of cats across the entirety of the islands. Another study on feral cat populations was performed by (Phillips et al., 2005) on Baltra Island, where feral cats have now been eradicated. The study did not measure direct effects of cats on the land iguana population there, it was on the methodology of the cat eradication program.

With over half the population of land iguanas exposed to cat predation, in depth studies on the cat populations that occur with land iguana populations would give an indication of the degree of threat the land iguanas are facing. Without the knowledge of the degree of threat it poses to land iguanas currently, the population trend of land iguanas cannot be uncovered. It is possible the land iguanas on Isabela Island are in major decline, however there is not data to support whether they are declining stable or increasing. At current there is minimal to no cat control on Isabela Island (K. Campbell, pers. comm), if this continues it is likely the cat population will grow in numbers and in fact may have already increased over the last decade, as they have the ability to breed profusely (Andersen, Martin & Roemer, 2004). An increase in the cat population without any form of effective control can be extremely damaging to the land iguana populations and renders the land iguana population on Isabela Island to be at serious risk of extinction.

Population estimates of feral cats across Isabela Island as well as determining the areas of the island where it is most populated with cats would provide a good starting point. Furthermore, population estimates of land iguanas in each sub-population would be beneficial and can be compared with that of the cat populations across the island. Surveying the populations for the occurrence of juvenile iguanas is recommended as low juvenile percentage is a strong indicator of cat depredation on juvenile land iguanas (Alberts et al., 2004).

Information of the cat populations and the measured effects they have on land iguanas in Isabela Island will firstly give insight on the current numbers of iguanas on Isabela Island as well as the sub-populations. Population trends could be uncovered and will contribute to a more informative IUCN Red List assessment of the species, whereby their risk of extinction and future trends can be determined accordingly.

Studies on variation in populations

Although some studies have been published on the variation in populations of land iguanas, they lack comprehensiveness and do not give reasons for what may influence these individual characteristics (Snell, Snell & Tracy, 1984). Additionally the variation has been explored more so in terms of the land iguana genetic makeup (Tzika et al., 2008).

Snell, Snell & Tracy (1984) discovered that land iguana populations display a varying range of SVL sizes, with small iguanas in South Plaza and large iguanas in Baltra Island, however did they not report or give an indication as to why their morphological characteristics may vary between islands. In Chapter 2 prior to our analysis I have displayed a summary table (table 2.2) of land iguanas from six islands. The table and the analysis clearly show there are significant differences between populations in length and mass, similar to Snell, Snell & Tracy's (1984) results. However instead, Isabela Island was found to have the largest iguanas in terms of length and mass.

Galapagos species are renowned for exhibiting high levels of morphological variation and have been researched extensively in the past years often leading to the discovery of new species and the realisation of a fast rate of speciation. Such species include Galapagos giant tortoises, Galapagos finches, Galapagos racer snakes, Lava lizards, Marine iguanas and more (Benavides et al., 2009; Burns, Hackett, & Klein, 2002; Grant, 1981; Jaffe, Slater, & Alfaro, 2011; Parent, Caccone, & Petren, 2008;

Zaher et al., 2018). Most recently in 2017, were five new subspecies of marine iguanas described (Miralles et al., 2017). It is possible that there are sub-species within *Conolophus subcristatus.* For instance Tzika et al. (2008) has speculated that iguanas from South Plaza may be a different species. This would have to be further investigated to confirm.

Marine iguanas have been more a subject of study for variation in size between islands. For instance Wikelski (2005) discovered that Marine iguanas were larger on western islands like Isabela and Fernandina and were smallest on eastern islands like Espanola and provides a few hypotheses for why this variation exists, such as predation, island area size, productivity in surrounding oceans and sexual selection influencing their body size and the variations that exist between the islands. More recently, Chiari et al. (2016) concluded that the size of marine iguanas and the differences in the island populations are due to sea surface temperature and more strongly island perimeter. Island perimeter or area size are both related factors and have now been mentioned twice as a major influence on the body size evolution of marine iguanas. Western islands such as Fernandina and Isabela Island are islands with larger area size (64,200 ha – 464,000 ha) and island perimeter than the central to eastern islands such as South Plaza and Espanola Island (13 ha – 6000 ha). Based on this previous research and our results it is likely the island size/perimeter plays an influence on body size of land iguanas.

The relationship of body size and island area is known as the 'island rule', this has been well documented in literature. Initially the island rule was investigated in mammals however research shows that the island rule appears to be exhibited for numerous species including birds, snakes and turtles, with larger individuals found in larger islands or in the mainland and smallest individuals on smaller islands (Lomolino, 1985, 2005). For instance, tri-coloured squirrels (*Callosciurus prevosti*) have shown to exhibit apparent variation in body length in Southeast Asian islands, with notably smaller body lengths in the smaller islands and larger body lengths in larger islands. Their sizes range from 198 mm to 274 mm (Heaney, 1978). Thus this could be the case for land iguanas.

Furthermore, although I did not record or quantify these features as such, I had noticed minor differences in the colouration and shape of the skull of land iguanas from different populations. For example, both sexes of land iguanas from the Fernandina population exhibited equally vibrant and deeply saturated yellow-orange coloration with slightly more elongated skulls as opposed to less vibrantly coloured individuals from the North Seymour population, especially the females being

significantly drabber in appearance (See figure 5.7). This is not to be interpreted as a fact but simply a side note of my observation. Further investigation is required to quantify the colour and shape variation between populations. Different colour morphs have been documented in other reptilian species such as blood pythons (*Python brongersma*), Dalmation wall lizards (*Podarcis melisellensis*) and even in Acklins rock iguanas (*Cyclura rileyi nuchalis*) (Huyghe et al., 2007; Iverson et al., 2015; Shine & Harlow, 1998).



Figure 5.7: Examples of male and female iguanas on Fernandina and North Seymour Island. Fernandina Island top left, female; bottom left: male. North Seymour, top right, female; bottom right, male. Notice individuals from Fernandina island show more saturation in yellow-orange pigment in scales (pigmentation not quantified in the field). Photos: K. Kumar

Further research on the differences between populations in phylogeny, genetics and morphology will be needed to determine the degree of environmental and genetic factors operating to produce the current variation observed. These studies can further the understanding of different land iguana populations and give insight on the environmental pressures that may be acting on them, allowing for increased successful management outcomes.

Eradicating feral cats

Currently there is no formal cat eradication campaign running in any of the islands in the Galapagos. The most recent campaign was on Baltra Island and ended in 2005 after successfully eradicating all cats from the island (Phillips et al., 2005). There are plans for an eradication campaign to commence in 2019 on Floreana Island. There has been minimal cat control initiated by the Galapagos National Park on Santa Cruz Island on Cerro Dragon for the protection of land iguanas (Campbell, K. pers. comm., 2018).

At least 48 islands have been eradicated completely of feral cats (Nogales et al., 2013) The eradication of feral cats has allowed a number of native species to recover. For instance the eradication of feral cats on sub-Antarctic Marion island in 1991 had led to increased breeding success of Great-winged (*Pterodroma macroptera*) and Blue Petrels (*Halobaena caerulea*), which had resulted in increased population growth (Cooper et al., 1995). Similarly many iguana species including Galapagos land iguanas have benefited greatly after the removal of cats from their home range (Alberts, Carter, & Hayes, 2004; Mitchell et al., 2002; Nogales et al., 2013; Phillips et al., 2005; Wilson, Robinson, & Vogel, 2004).

To ensure the long-term survival of land iguanas, I recommend implementing strong cat control programmes throughout Isabela Island. Eradicating cats or reducing their numbers on Isabela Island and Santa Cruz Island would be removing the primary threat to Galapagos land iguanas. The removal of cats will also benefit a number of other native species that are also effected by cat predation (Phillips, Wiedenfeld & Snell, 2012).

The cat eradication on Baltra Island is an example of a successful eradication that has shown to have positive effects on the land iguana population there (Phillips et al., 2005). Baltra is of relatively small size so the complete eradication of cats was feasible. On the contrary, Isabela Island is large and generally would not be considered as feasible to completely eradicate all cats from the island (Medina et al., 2011; Nogales et al., 2013). Nevertheless, a reduction in numbers and persistent and ongoing cat control will have positive effects on the native wildlife including the land iguanas. Targeting the core areas of land iguana populations for restoration and eradication should be prioritized, once the core areas have been identified.

Population Viability Analysis (PVA)

Current data available on Galapagos land iguanas is not sufficient to make solid projections on their populations trends. I have mentioned previously an estimated decline of land iguana populations in Isabela island due to the presence of cats. This however has never been measured, thus the scale of the threat is unknown as well as the current trends of the land iguana population. In order to make precise and accurate estimates that are specific in number and give an indication of a time frame, a Population Viability Analysis (PVA) would prove as a useful tool.

PVA models are valuable tools in conservation biology used for endangered or threatened species (Beissinger & Westphal, 1998). PVA's are based on computer simulations. Several different models and computer software are available depending on the species of interest and the circumstances. The simulation produces predicted future projections for the population of interest. Ultimately it is used to assess the risk of extinction or decline for the population and can provide the probability of decline as well an estimation of the timeframe within which the population may experience decline or extinction (Keedwell, 2004). A PVA can identify specific threats and their individual effects on a population which can then consequently be used to evaluate management options. Furthermore, it can help with prioritizing different needs and research for the species or population of interest (Reşit Akçakaya & Sjögren-Gulve, 2000).

A PVA can also be used to model translocation scenarios for a species. For instance, a PVA was conducted on four different populations of Caribou in the Canadian Rocky Mountains to assess the viability of translocation recovery of this species. The result of the PVA show that one population located in Banff within the Canadian Rocky Mountains have a 53-98% chance of over eight females remaining after 20 years once translocated, this suggests that translocations are in fact a suitable mode of recovery for this particular population. As for the other three populations the PVA indicated that two of the populations would not benefit from a translocation and would not be sufficient enough to prevent their extinction and that one of the populations did not require a translocation and would remain a viable population without being translocated (Decesare et al., 2011). PVA's thus prove to be useful tools for planning translocations and assessing modes of conservation specific to the species. Similarly a PVA can be used as a post-release monitoring method to ensure the success of translocations and assess the spread or establishment of the founder population. For example, newly translocated populations are

often initially small as this is a result of the limited funding available for the project. Small populations are vulnerable to environmental stochasticity and the Allee effect, thus their long-term persistence may be restricted by their size (Deredec & Courchamp, 2007). A PVA can be used to model projections of the newly translocated population and give an insight into the persistence of the population and when it is appropriate to introduce more individuals into a population or what conservation measures are required to ensure the persistence of a population. Nolet & Baveco (1996) conducted a PVA as part of a post-monitoring programme following a beaver (*Castor fiber*) translocation in the Netherlands and found that the population is not viable unless the reproductive success of the population increases. These findings allowed Nolet & Baveco (1996) to identify the limitation that affect the viability of the population of translocated beavers (reproductive success), this then provided them with an incentive to look into the next possible management strategies for the population with a focus on reproductive success.

Performing a PVA on land iguanas would provide as a useful tool for identifying current and future trends. It can also help to identify future populations that are suitable for translocations and also provide as a post-release measure following the translocation of land iguanas from North Seymour to Santiago Island. Performing a PVA will require extensive data on the populations of interest, a large and comprehensive dataset will benefit the output of the PVA as if the data are richer, then consequently the PVA can be expected to produce quality estimates (Coulson et al., 2001). Although the results of a PVA, should still be treated as an estimate. A PVA would fill the current gaps of knowledge in land iguana research and will help to identify immediate threats faced by land iguanas as well as the degree of threat. The level of tolerance to a specific threat such as cat predation can be identified to uncover how much a population can withstand before it experiences decline (Laufenberg, Clark, & Chandler, 2018). This will better inform future management decisions and assist the direction of future research and monitoring for land iguanas in the Galapagos.

MaxEnt: modelling species distribution

Maximum Entropy modelling (MaxEnt) was considered as a key component at the onset of this study. Our aim was to model potential land iguana distribution on Santiago Island prior to the translocation. Specifically we wanted to reveal the time taken for Santiago Island to reach a point of

saturation, whereby land iguanas are distributed across the entire habitable range of the island. However, this had to be abandoned due to the difficulty of acquiring environmental layers that are needed for this form of modelling (Townsend, Papeş, & Eaton, 2007).

MaxEnt is one of the main and most common computer programs used for modelling species distributions (Elith et al., 2011). It can be used to inform and predict several conservation scenarios. For instance it has been used to predict current distributions of the Violet-throated Metaltail hummingbird in the Andes to assess its extinction risk (Tinoco et al., 2009). Alternatively, Ward (2007) used MaxEnt modelling to predict the potential distribution of six invasive ant species in New Zealand, this study used the environmental layer of climate and found that coastal and lowland areas of New Zealand are suitable for the six invasive ant species, consequently revealing their potential future range.

Currently, the range of certain population of land iguanas, for example those on Isabela Island, are poorly studied. It is unknown exactly how much of the island land iguanas occupy. This is also the same case for Fernandina Island, where the habitat that land iguanas occupy on Fernandina Island has never been quantified. Our research in Chapter 3 is the only study that aims to quantify land iguana range on Fernandina Island with the use of vegetation maps provided by from: http://institutodegeografia.org/vega-2/ , as shown in figure 3.5. This is however not a robust enough estimate as this was quantified purely through the vegetation map and past observations of land iguanas in certain 'preferable' habitats that was mentioned in earlier land iguana studies (Snell, Snell, & Tracy, 1984; Werner, 1983).

MaxEnt would provide beneficial for modelling the current distribution of land iguanas on the islands they occupy, namely Isabela Island. The results can then help to increase the accuracy of population estimates as well as revealing the area of occupancy of land iguanas. This can contribute to a thorough overview of the range of land iguanas in the islands they occupy, concurrently this can help to better quantify their extinction risk, as given by the IUCN Red List Assessment (Syfert et al., 2014) under 5b (Data for criterion B: restricted range), see Chapter 4. MaxEnt can also be used as mentioned before, to predict the potential distribution of land iguanas on Santiago Island. The simulations produced by MaxEnt can give insights on the time it takes for land iguanas to saturate their habitat as well as how much of the habitat in Santiago can be potentially occupied by land iguanas. For this, data on environmental layers of the islands of interest will be needed as well as presence data from land iguanas

(Townsend, Papeş, & Eaton, 2007). This can also be used as a post-monitoring measure following the translocation of land iguanas to Santiago Island, whereby tracking their dispersal on the island as well as assessing habitat suitability (Tinoco et al., 2009). Lastly MaxEnt modelling can help to identify several suitable future translocation sites for land iguanas on Santiago Island (Chauvenet, 2013).

REFERENCES

Adsersen, H. (1989). The rare plants of the Galápagos Islands and their conservation. *Biological Conservation, 47*(1), 49-77.

AFP. (2015). Mexican arrested for trying to smuggle iguanas from Galapagos Islands. *The Telegraph*.

Andersen, M. C., Martin, B. J., & Roemer, G. W. (2004). Use of matrix population models to estimate the efficacy of euthanasia versus trap-neuter-return for management of free-roaming cats. *Journal of the American Veterinary Medical Association*, 225(12), 1871-1876.

Akçakaya, H., & Ginzburg, L. (1991). Ecological risk analysis for single and multiple populations *Species conservation: a population-biological approach* (pp. 73-87): Springer.

Akçakaya, H. R., Mills, G., & Doncaster, C. P. (2007). The role of metapopulations in conservation. *Key topics in conservation biology*, 64-84.

Akçakaya, H. R., & Sjögren-Gulve, P. (2000). Population viability analyses in conservation planning: an overview. *Ecological Bulletins*, 9-21.

Alberts, A. C. (2007a). Behavioral considerations of headstarting as a conservation strategy for endangered Caribbean rock iguanas. *Applied Animal Behaviour Science, 102*(3-4), 380-391.

Alberts, A. C. (2007b). Behavioral considerations of headstarting as a conservation strategy for endangered Caribbean rock iguanas. *Applied Animal Behaviour Science, 102*, 380-391.

Alberts, A. C., Carter, R. L., & Hayes, W. K. (2004). *Iguanas: Biology and Conservation*: Berkeley : University of California Press, 2004.

Alberts, A. C., Worley, M. B., Oliva, M. L., Telford Jr, S. R., Morris, P. J., & Janssen, D. L. (1998). The need for pre-release health screening in animal translocations: A case study of the Cuban iguana (*Cyclura nubila*). *Animal Conservation, 1*(3), 165-172.

Andel, J. v., & Aronson, J. (2012). *Restoration Ecology: the New Frontier*: Hoboken : John Wiley & Sons.

Anderson, D. R., Laake, J. L., Crain, B. R., & Burnham, K. P. (1979). Guidelines for line transect sampling of biological populations. *The Journal of Wildlife Management*, 70-78.

Anderson, D. P., McMurtrie, P., Edge, K. A., Baxter, P. W. J., & Byrom, A. E. (2016). Inferential and forward projection modeling to evaluate options for controlling invasive mammals on islands. *Ecological Applications, 26*(8), 2546-2559.

Aplasca, A. C., Hekkala, E. R., Iverson, J. B., Welch, M. E., & Colosimo, G. (2016). Genetic diversity and structure in the Endangered Allen Gays Rock Iguana, *Cyclura cychlura inornata*. *PeerJ, 4*.

Armstrong, D., Hayward, M. W., Moro, D., & Seddon, P. J. (2015). *Advances in reintroduction biology of Australian and New Zealand fauna*: Clayton South, VIC : CSIRO Publishing, [2015].

Armstrong, D. P., & Davidson, R. S. (2006). Developing population models for guiding reintroductions of extirpated bird species back to the New Zealand mainland. *New Zealand Journal of Ecology, 30*(1), 73-85.

Atkinson, I. A., & Cameron, E. K. (1993). Human influence on the terrestrial biota and biotic communities of New Zealand. *Trends In Ecology & Evolution, 8*(12), 447-451.

August, P. (1985). Islands of Mytilus as a habitat for small intertidal animals: effect of island size on community structure·. *Marine Ecology, 25*, 71-81.

Baling, M., Stuart-Fox, D., Brunton, D. H., & Dale, J. (2016). Habitat suitability for conservation translocation: The importance of considering camouflage in cryptic species. *Biological Conservation, 203*, 298-305.

Balme, G. A., Slotow, R., & Hunter, L. T. (2009). Impact of conservation interventions on the dynamics and persistence of a persecuted leopard (*Panthera pardus*) population. Biological Conservation, 142(11), 2681-2690.

Barnett, D. B., & Rudd, L. R. (1983). Feral dogs of the Galapagos islands: Impact and control. *Animal Studies Repository, 4*.

Barnett, S. (1985). *New Zealand in the Wild*: An Illustrated AZ of Native and Introduced Birds, Mammals, Reptiles and Amphibians. Collins.

Barnosky, A. D., Matzke, N., Tomiya, S., Wogan, G. O. U., Swartz, B., Quental, T. B., Ferrer, E. A. (2011). Has the Earth's sixth mass extinction already arrived? *Nature, 471*(7336), 51-57.

Barun, A., Hanson, C. C., Campbell, K. J., & Simberloff, D. (2011). A review of small Indian mongoose management and eradications on islands. *Island Invasives: Eradication and Mangement, IUCN*, 17-25.

Batson, W. G., Gordon, I. J., Manning, A. D., & Fletcher, D. B. (2015). Translocation tactics: A framework to support the IUCN Guidelines for wildlife translocations and improve the quality of applied methods. *Journal of Applied Ecology, 52*(6), 1598-1607.

Begon, M. (1979). *Investigating Animal Abundance: capture-recapture for biologists*: Edward Arnold Ltd.

Beissinger, S. R., & Westphal, M. I. (1998). On the use of demographic models of population viability in endangered species management. *The Journal of Wildlife Management*, 821-841.

Bell, T. P., & Herbert, S. M. (2017). Establishment of a self-sustaining population of a long-lived, slow-breeding gecko species (*Diplodactylidae: Hoplodactylus duvaucelii*) evident 15 years after translocation. *Journal of Herpetology, 51*(1), 37-46.

Bellingham, P. J., Towns, D. R., Cameron, E. K., Davis, J. J., Wardle, D. A., Wilmshurst, J. M., & Mulder, C. P. H. (2010). New Zealand island restoration: seabirds, predators, and the importance of history. *New Zealand Journal of Ecology*(1), 115.

Benavides, E., Baum, R., Snell, H. M., Snell, H. L., & Sites, J., Jack W. (2009). Island biogeography of Galápagos lava lizards (*Tropiduridae: Microlophus*): species diversity and colonization of the archipelago. *Evolution: International Journal of Organic Evolution, 63*(6), 1606-1626.

Biggs, A. L. (1990). Coevolution in the Galápagos: An example for the biology classroom. *The American Biology Teacher, 52*(1), 24-28.

Bland, L. M., & Böhm, M. (2016). Overcoming data deficiency in reptiles. *Biological Conservation, 204*(Part A), 16-22.

Bland, L. M., Collen, B., Orme, C. D. L., & Bielby, J. (2015). Predicting the conservation status of data-deficient species. *Conservation Biology*(1), 250.

Bock, C. B., Paez, P. V., Rand, S. A., & Burghardt, M. G. (2016). Life table and stochastic matrix projection analysis for a population of green iguanas (*Iguana iguana*): implications for conservation and control. *Herpetological Conservation and Biology, 11*, 47-60.

Böhm, M., Collen, B., Baillie, J. E. M., Bowles, P., Chanson, J., Cox, N., Zug, G. (2013). The conservation status of the world's reptiles. *Biological Conservation, 157*, 372-385.

Booy, G., Hendriks, R., Smulders, M., Van Groenendael, J., & Vosman, B. (2000). Genetic diversity and the survival of populations. *Plant biology, 2*(04), 379-395.

Brook, B. W., Sodhi, N. S., & Ng, P. K. L. (2003). Catastrophic extinctions follow deforestation in Singapore. *Nature, 424*(6947), 420.

Buckland, S., Cole, N. C., Godsall, B., Rodríguez-Pérez, J., Gallagher, L. E., Henshaw, S. M., & Harris, S. (2014). Original research article: Habitat selection of the Mauritian lowland forest day gecko at multiple spatial scales: A baseline for translocation. *Global Ecology and Conservation, 1*, 71-79.

Buckland, S. T. (2001). *Introduction to distance sampling : estimating abundance of biological populations*: Oxford ; New York : Oxford University Press, 2001.

Buckland, S. T., Anderson, D. R., Burnham, K. P., Laake, J. L., Borchers, D. L., & Thomas, L. (2004). *Advanced Distance Sampling: Estimating abundance of biological populations*: Oxford ; New York : Oxford University Press, 2001.

Buckley, L., de Queiroz, K., Grant, T., Hollingsworth, B., Iverson, J., Pasachnik, S., & Stephen, C. (2016). A checklist of the iguanas of the world (*Iguanidae; Iguaninae*). *Herpetological Conservation & Biology, 11*(6), 4-46.

Buckley, L., Etheridge, R., Hollingsworth, B., & Malone, C. (2011). *Iguanas of the world*.

Burger, J., & Gochfeld, M. (1991). Burrow site selection by black iguana (*Ctenosaura similis*) at Palo Verde, Costa Rica. *Journal of Herpetology*, 430-435.

Burghardt, G. M., Greene, H. W., & Rand, A. S. (1977). Social behavior in hatchling green iguanas: life at a reptile rookery. *Science, 195*(4279), 689-691.

Burghardt, G. M., & Rand, S. A. (1982). *Iguanas of the world: their behaviour, ecology and conservation*.

Burnham, K. P., & Anderson, D. R. (1992). Data-based selection of an appropriate biological model: the key to modern data analysis *Wildlife 2001: populations* (pp. 16-30): Springer.

Burns, K. J., Hackett, S. J., & Klein, N. K. (2002). Phylogenetic relationships and morphological diversity in Darwin's finches and their relatives. *Evolution, 56*(6), 1240-1252.

Burton, F. J., & Rivera-Milan, F. F. (2014). Monitoring a population of translocated Grand Cayman blue iguanas: assessing the accuracy and precision of distance sampling and repeated counts. *Animal Conservation, 17*, 40-47.

Burton, F. J., & Rivera-Milán, F. F. (2014). Monitoring a population of translocated Grand Cayman blue iguanas: assessing the accuracy and precision of distance sampling and repeated counts. *Animal Conservation, 17*, 40-47.

Campbell, L. M. (2012). Seeing red : inside the science and politics of the iucn red list. *Conservation and Society*(4), 367.

Cardinale, B. J., Duffy, E. J., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature*(7401), 59.

Carpenter, C. C. (1969). Behavioral and ecological notes on the galapagos land iguanas. *Herpetologists' League,* 155.

Carrion, V., Donlan, C. J., Campbell, J. K., Lavoie, C., & Cruz, F. (2011). Archipelago-wide island restoration in the galapagos islands: reducing costs of invasive mammal eradication programs and reinvasion risk. *Plos One, 6*.

Carrion, V., Sevilla, C., & Tapia, W. (2008). Management of introduced animals in galapagos. *Galapagos Research, 65*, 46-48.

Case, T. J., Bolger, D. T., & Richman, A. D. (1992). Reptilian extinctions: the last then thousand years. *Conservation Biology*.

Cayot, J. L., Menoscal, R. (1992). Land iguanas return to Baltra. *Noticias De Galapagos*, 11-13.

Cayot, J. L. (2008). The restoration of giant tortoise and land iguana populations in galapagos. *Journal of Science and Conservation in the Galapagos Islands*(65), 39-43.

Ceballos, G., Ehrlich, P. R., Barnosky, A. D., Garcia, A., Pringle, R. M., & Palmer, T. M. (2015). Accelerated modern human-induced species losses: Entering the sixth mass extinction. *Science Advances, 1*(5).

Ceballos, G., Ehrlich, P. R., & Dirzo, R. (2017). Biological annihilation via the ongoing sixth mass extinction signaled by vertebrate population losses and declines. *Proceedings of the National Academy of Sciences, 114*(30), E6089-E6096.

Ceballos, G., García, A., & Ehrlich, P. R. (2010). The sixth extinction crisis loss of animal populations and species. *Journal of Cosmology, 8*, 1821-1831.

Chapman, D. G., & Overton, W. S. (1966). Estimating and testing differences between population levels by the schnabel estimation method. *The Journal of Wildlife Management*(1), 173.

Chauvenet, A. L. M. (2013). *Modelling the dynamics of translocated populations.* Imperial College London, in collaboration with the Institute of Zoology of the Zoological Society of London.

Cheyne, S. M. (2006). Wildlife reintroduction: considerations of habitat quality at the release site. *BMC Ecology, 6*(1), 5. doi:10.1186/1472-6785-6-5

Chiari, Y., Glaberman, S., Tarroso, P., Caccone, A., & Claude, J. (2016). Ecological and evolutionary influences on body size and shape in the Galápagos marine iguana (*Amblyrhynchus cristatus*). *Oecologia, 181*(3), 885-894.

Christian, K. A., Tracy, C. R., & Porter, W. P. (1984). Diet, digestion, and food preferences of galapagos land iguanas. *Herpetologists' League, 40*(2), 205-212.

Clark, D. A. (1981). Foraging patterns of black rats across a desert-montane forest gradient in the Galapagos Islands. *Biotropica*, 182-194.

Clark, J. A., & May, R. M. (2002). Taxonomic bias in conservation research. *Science*(5579), 191.

Claudino-Sales, V. (2019). Galápagos Islands, Ecuador *Coastal World Heritage Sites* (pp. 327-333): Springer.

Cooper, J., Marais, A., Bloomer, J., & Bester, M. (1995). A success story: Breeding of burrowing petrels (*Procellariidae*) before and after the eradication of feral cats *Felis catus* at subantarctic Marion Island. *Marine Ornithology, 23*(1), 33-37.

Costantini, D., Dell'Omo, G., Casagrande, S., Fabiani, A., Carosi, M., Bertacche, V., Gentile, G. (2005). Inter-population variation of carotenoids in Galápagos land iguanas (*Conolophus subcristatus*). *Comparative Biochemistry and Physiology, Part B, 142*, 239-244.

Costantini, D., Dell'Omo, G., De Filippis, Stefania P., Marquez, C., Snell, Howard L., Snell, Heidi M., Gentile, G. (2009). Temporal and spatial covariation of gender and oxidative stress in the galápagos land iguana *Conolophus subcristatus*. *Physiological and Biochemical Zoology 82,* 430-437.

Costello, M. J., May, R. M., & Stork, N. E. (2013). Can we name Earth's species before they go extinct? *Science*(6118), 413.

Coulson, T., Mace, G. M., Hudson, E., & Possingham, H. (2001). The use and abuse of population viability analysis. *Trends In Ecology & Evolution, 16*(5), 219-221.

Creel, S., Winnie Jr, J., Maxwell, B., Hamlin, K., & Creel, M. (2005). Elk alter habitat selection as an antipredator response to wolves. *Ecology, 86*(12), 3387-3397.

Cruz, F., Carrion, V., Campbell, K. J., Lavoie, C., & Donlan, C. J. (2009). Bio-economics of large-scale eradication of feral goats from santiago island, galápagos. *The Journal of Wildlife Management*(2), 191.

Cruz, F., Donlan, C. J., Campbell, K., & Carrion, V. (2005). Conservation action in the Galapagos: feral pig (*Sus scrofa*) eradication from Santiago Island. *Biological Conservation, 121*(3), 473-478.

Dagmar, I. W. (1988). The effect of varying water potential on body weight, yolk and fat bodies in neonate green iguanas. *Copeia*(2), 406.

Daniel, L., & Berovides, V. (2004). Burrows and morphology of Cuban iguanas (*Cyclura nubila*) inhabiting Cruz del Padre Cays, Sabana-Camagüey Archipelago, North of Matanzas. *Iguana Specialist Group Newsletter, 7*(1), 7-9.

Darwin, Charles (1845). *The voyage of the Beagle*.

David, L. H., Qijian, S., Youlin, Z., Ik-Young, C., Randall, L. N., Jose, M. C., Perry, B. C. (2006). Impacts of genetic bottlenecks on soybean genome diversity. *Proceedings of the National Academy of Sciences of the United States of America*(45), 16666.

de la Navarre, B. J. S. (2013). Herpetology: An Introductory Biology of Amphibians and Reptiles (4th edition). *Journal of the American Veterinary Medical Association, 243*(12), 1710-1711.

de los Ríos, C., Watson, J., & Butt, N. (2018). Persistence of methodological, taxonomical, and geographical bias in assessments of species' vulnerability to climate change: A review. *Global Ecology and Conservation, 15*, e00412.

De Vos, J. M., Joppa, L. N., Gittleman, J. L., Stephens, P. R., & Pimm, S. L. (2015). Estimating the normal background rate of species extinction. *Conservation Biology*(2), 452.

Decesare, N. J., Whittington, J., Hebblewhite, M., Robinson, H., Bradley, M., Neufeld, L., & Musiani, M. (2011). The role of translocation in recovery of woodland caribou populations. *Conservation Biology, 25*(2), 365-373.

Deredec, A., & Courchamp, F. (2007). Importance of the Allee effect for reintroductions. *Ecoscience, 14*(4), 440-451.

Derek, S., Barry, W. B., Richard, F., & Barbara, A. S. (2004). Most species are not driven to extinction before genetic factors impact them. *Proceedings of the National Academy of Sciences of the United States of America*(42), 15261.

Dickman, A., & Marker, L. (2005). Factors affecting leopard (*Panthera pardus*) spatial ecology, with particular reference to Namibian farmlands. South African Journal of Wildlife Research-24-month delayed open access, 35(2), 105-115.

Dirzo, R., & Raven, P. H. (2003). Global state of biodiversity and loss. *Annual Review of Environment & Resources, 28*(1), 137.

Dodd, C. K. J., & Seigel, R. A. (1991). Relocation, repatriation, and translocation of amphibians and reptiles: are they conservation strategies that work? *Herpetologica*, 336-350.

Ducarme, F., Luque, G. M., & Courchamp, F. (2013). What are "charismatic species" for conservation biologists. *BioSciences Master Reviews*, 10(2013), 1-8.

Ducatez, S., & Shine, R. (2017). Drivers of extinction risk in terrestrial vertebrates. *Conservation Letters, 10*(2), 186-194.

Duméril, C., Bibron, G., & Duméril, A. H. A. (1839). *Erpétologie générale: ou, Histoire naturelle complète des reptiles* (Vol. 5): Roret.

Duque, A. B., Taphorn, D. C., & Winemiller, K. O. (1998). Ecology of the coporo, *Prochilodus mariae* (Characiformes, Prochilodontidae), and status of annual migrations in western Venezuela. *Environmental Biology of Fishes*(1), 33.

Eckhardt, R. C. (1972). Introduced plants and animals in the Galapagos Islands. BioScience, 22(10), 585-590.

Ellis, H. I., & James P, R. (1978). Field observations of cooling rates of galapagos land iguanas (*Conolophus subcristatus*). *Comparative Biochemistry and Physiology -- Part A: Physiology*, 59(2), 205-209.

Elith, J., Phillips, S. J., Hastie, T., Dudík, M., Chee, Y. E., & Yates, C. J. (2011). A statistical explanation of MaxEnt for ecologists. *Diversity and distributions, 17*(1), 43-57.

Evans, F. C. (1951). Notes on a population of the striped ground squirrel (*Citellus tridecemlineatus*) in an abandoned field in Southeastern Michigan. *Journal of Mammalogy*(4), 437.

Fabiani, A., Trucchi, E., Rosa, S., Marquez, C., Snell, H. L., Snell, H. M., . . . Gentile, G. (2011). Conservation of Galapagos land iguanas: genetic monitoring and predictions of a long-term program on the island of Santa Cruz. *Animal Conservation, 14*(4), 419-429.

Falcón, W., Ackerman, J. D., Recart, W., & Daehler, C. C. (2013). Biology and Impacts of Pacific Island Invasive Species. 10. *Iguana iguana*, the Green Iguana (*Squamata: Iguanidae*)1. *Pacific Science*(2), 157. doi:10.2984/67.2.2

Fitzinger, L.-J. (1843). *Systema reptilium* (Vol. 1): Braumüller & Seidel.

Fortin, D., Beyer, H. L., Boyce, M. S., Smith, D. W., Duchesne, T., & Mao, J. S. (2005). Wolves influence elk movements: behavior shapes a trophic cascade in Yellowstone National Park. *Ecology, 86*(5), 1320-1330.

Foufopoulos, J., Kilpatrick, A. M., & Ives, A., R. (2011). Climate change and elevated extinction rates of reptiles from mediterranean islands. *The American Naturalist*(1), 119.

Franco, A., Hendriksen, R. S., Lorenzetti, S., Onorati, R., Gentile, G., Dell'Omo, G., . . . Battisti, A. (2011). Characterization of salmonella occurring at high prevalence in a population of the land iguana *Conolophus subcristatus* in Galapagos islands, Ecuador. *Plos One*, 6(8).

Frankham, R. (1995). Conservation genetics. *Annual Review of Genetics, 29*, 305-327.

Frankham, R. (1998). Inbreeding and extinction: island populations. *Conservation Biology*(3), 665.

Frankham, R. (2010). Challenges and opportunities of genetic approaches to biological conservation. *Biological Conservation, 143*(9), 1919-1927.

Freake, M. J., & DePerno, C. S. (2017). Importance of demographic surveys and public lands for the conservation of eastern hellbenders *Cryptobranchus alleganiensis alleganiensis* in southeast USA. *Plos One, 12*(6), 1-16.

Fritts, T. H. (1984). Evolutionary divergence of giant tortoises in Galapagos. *Biological Journal of the Linnean Sociey, 21*(1-2), 165-176.

Funk, W. C., Almeida-Reinoso, D., Nogales-Sornosa, F., & Bustamante, M. R. (2003). Monitoring population trends of *Eleutherodactylus* frogs (Vol. 37, pp. 245-256).

Galliard, J.-F. o. L., Fitze, P., S., Ferrière, R., & Clobert, J. (2005). Sex ratio bias, male aggression, and population collapse in lizards. *Proceedings of the National Academy of Sciences of the United States of America*(50), 18231.

Gaywood, M. J. (2018). Reintroducing the Eurasian beaver *Castor fiber* to Scotland. *Mammal Review, 48*(1), 48-61.

Gentile, G., Ciambotta, M., & Tapia, W. (2013). llegal wildlife trade in Galapagos: molecular tools helpthe taxonomic identification of confiscated iguanas and guidetheir rapid repatriation. *Conservation Genetics Resources, 5*(3), 867-872.

Gentile, G., Fabiani, A., Marquez, C., Snell, H. L., Snell, H. M., Tapia, W., . . . Ayala, F. J. (2009). An overlooked pink species of land iguana in the galápagos. *PNAS, 106,* 507-511.

Gentile, G., Marquez, C., Snell, H. L., Tapia, W., & Izurieta, A. (2016). Conservation of a New Flagship Species: The Galápagos Pink Land Iguana. *Problematic Wildlife, 315-336*

Gentile, G., & Snell, H. (2009). *Conolophus marthae* sp.nov. (*Squamata, Iguanidae*), a new species of land iguana from the Galápagos archipelago. *Zootaxa*(2201), 1-10.

Germano, J. M., & Bishop, P. J. (2009). Suitability of amphibians and reptiles for translocation. *Conservation Biology, 23*(1), 7-15.

Gese, E. M. (2001). Monitoring of terrestrial carnivore populations. *USDA National Wildlife Research Center - Staff Publications, 576*.

Gibbs, J. P., Marquez, C., & Sterling, E. J. (2008). The role of endangered species reintroduction in ecosystem restoration: tortoise–cactus interactions on Española Island, Galápagos. *Restoration Ecology, 16*(1), 88-93.

Gillespie, G. (2010). Population age structure of the spotted tree frog (*Litoria spenceri*): insights into population declines (Vol. 37, pp. 19-26).

Goodyear, N. C., & Lazell, J. (1994). Status of a relocated population of endangered *Iguana pinguis* on Guana Island, British Virgin Islands. *Restoration Ecology, 2*(1), 43-50.

Grant, P. R. (1981). Speciation and the adaptive radiation of Darwin's Finches: the complex diversity of Darwin's finches may provide a key to the mystery of how intraspecific variation is transformed into interspecific variation. *American Scientist, 69*(6), 653-663.

Gray, J. E. (1831). Description of a new species of *Amblyrhynchus* of Mr. Bell, in the British Museum. *Zoological Miscellany, 10*(6).

Gray, J. E. (1845). *Catalogue of the Specimens of Lizards in the Collection of the British Museum*: order of the Trustees.

Grayson, K. L., Mitchell, N. J., Monks, J. M., Keall, S. N., Wilson, J. N., & Nelson, N. J. (2014). Sex ratio bias and extinction risk in an isolated population of Tuatara (*Sphenodon punctatus*). *Plos One, 9*(4), 1-10.

Griffith, B., Scott, J. M., Carpenter, J. W., & Reed, C. (1989). Translocation as a species conservation tool: status and strategy. *Science*(4917), 477.

Griffith, B., Scott, M. J., Carpenter, J. W., & Reed, C. (1993). Animal translocations and potential disease transmission. *Journal of Zoo and Wildlife Medicine*(3), 231.

Grimm, A., Gruber, B., & Henle, K. (2014). Reliability of different mark-recapture methods for population size estimation tested against reference population sizes constructed from field data. *Plos One, 9*(6).

Groombridge, J. J., Massey, J. G., Bruch, J. C., Malcolm, T., Brosius, C. N., Okada, M. M., ... & VanderWerf, E. A. (2004). An attempt to recover the Po'ouli by translocation and an appraisal of recovery strategy for bird species of extreme rarity. *Biological Conservation*, 118(3), 365-375.

Hall, J., & Fleishman, E. (2010). Demonstration as a means to translate conservation science into practice. *Conservation Biology, 24*(1), 120-127.

Hamann, O. (1979). On climatic conditions, vegetation types, and leaf size in the Galapagos Islands. *Biotropica*, 101-122.

Harlow, P. S., & Biciloa, P. N. (2001). Abundance of the Fijian crested iguana (*Brachylophus vitiensis*) on two islands. *Biological Conservation, 98*(2), 223-231.

Harnik, P. G., Lotze, H. K., Anderson, S. C., Finkel, Z. V., Finnegan, S., Lindberg, D. R., Tittensor, D. P. (2012). Extinctions in ancient and modern seas (Vol. 27, pp. 608-617).

Harper, G., & Carrion, V. (2011). Introduced rodents in the Galápagos: colonisation, removal and the future. Isl. Invasives Erad. Manag, 63-66.

Harper, G. A., Zabala, J., & Carrion, V. (2011). Monitoring of a population of Galápagos land iguanas (*Conolophus subcristatus*) during a rat eradication using brodifacoum. *Island Invasives: Eradication and Mangement*, 309-312.

Harris, D., & Macdonald, D. (2007). Interference competition between introduced black rats and endemic Galapagos rice rats. *Ecology, 88*(9), 2330-2344.

Heaney, L. (1978). Island area and body size of insular mammals: evidence from the tri-colored squirrel (*Callosciurus prevosti)* of Southeast Asia. *Evolution, 32*(1), 29-44.

Heckel, D. G., & Roughgarden, J. (1979). A technique for estimating the size of lizard populations. *The Ecological Society of America, 60*(5), 966-975.

Hedrick, P. W., Lacy, R. C., Allendorf, F. W., & Soule, M. E. (1996). Directions in conservation biology: comments on Caughley. *Conservation Biology*(5), 1312.

Heleno, R., Blake, S., Jaramillo, P., Traveset, A., Vargas, P., & Nogales, M. (2011). Frugivory and seed dispersal in the Galápagos: what is the state of the art? *Integrative Zoology, 6*(2), 110-129.

Heller, E. (1903). *Reptiles* (Vol. 14): The Academy.

Helsen, P., Browne, R. A., Anderson, D. J., Verdyck, P., & Van Dongen, S. (2009). Galapagos' *Opuntia* (prickly pear) cacti: extensive morphological diversity, low genetic variability. *Biological Journal of the Linnean Society, 96*(2), 451-461.

Henderson, R. W. (1992). Consequences of predator introductions and habitat destruction on amphibians and reptiles in the post-Columbus West Indies *Caribbean Journal of Science, 28*(1), 1-10.

Hendrix, L. B. (1981). Post-eruption succession on Isla Fernandina, Galapagos. *Madrono, 28*(4), 242-254.

Herbert, W. J. (1989). Predation and the mediation of intraspecific competition in an infaunal community in the Bay of Fundy. *Journal of experimental marine biology and ecology*, 132(3), 221-245.

Hickman, C. P., Jr., Roberts, L. S., Eisenhour, D. J., Keen, S. L., Larson, A., Ober, W. C., & Garrison, C. W. (2015). *Animal diversity*: New York, NY McGraw-Hill Education, [2015]. Seventh edition.

Hilderbrand, R., Watts, A., & Randle, A. (2005). The myths of restoration ecology. *Ecology and Society, 10*(1).

Hite, J. L., Gómez, C. A. R., Larimer, S. C., Díaz-Lameiro, A. M., & Powell, R. (2008). Anoles of St. Vincent (*Squamata: Polychrotidae*): Population densities and structural habitat use. *Caribbean Journal of Science*(1), 102.

Hoffmann, M., Brooks, T., Da Fonseca, G., Gascon, C., Hawkins, A., James, R., . . . Rodrigues, A. (2008). Conservation planning and the IUCN Red List. *Endangered Species Research, 6*(2), 113-125.

Hofkin, V. B., Wright, A., Altenbach, J., Rassmann, K., Snell, M. H., Miller, D. R., . . . Snell, L. H. (2003). Ancient DNA gives green light to Galápagos Land Iguana repatriation. *Conservation Genetics, 4*, 105.

Honegger, R. (1981). List of amphibians and reptiles either known or thought to have become extinct since 1600. *Biological Conservation, 19*(2), 141-158.

Howald, G., Donlan, C. J., Gálvan, J. P., Russell, J. C., Parkes, J., Samaniego, A., . . . Tershy, B. (2007). Invasive rodent eradication on islands. *Conservation Biology*(5), 1258.

Howard, S. D., & Bickford, D. P. (2014). Amphibians over the edge: silent extinction risk of Data Deficient species. *Diversity & Distributions, 20*(7), 837-846.

Hubbs, A. H., Karels, T., & Boonstra, R. (2000). Indices of population size for burrowing mammals. *The Journal of Wildlife Management*, 296-301.

Hughes, J., Daily, G. C., & Ehrlich, P. R. (1997). Population diversity: its extent and extinction. *Science*(5338), 689.

Huyghe, K., Vanhooydonck, B., Herrel, A., Tadić, Z., & Van Damme, R. (2007). Morphology, performance, behavior and ecology of three color morphs in males of the lizard Podarcis melisellensis. *Integrative and Comparative Biology, 47*(2), 211-220.

Isaac, N., & Cowlishaw, G. (2004). How species respond to multiple extinction threats. *Proceedings of the Royal Society B: Biological Sciences, 271*(1544), 1135.

IUCN. (1994). *1994 Categories and Criteria (version 2.3)*. Gland, Switzerland.

IUCN. (2017). *Guidelines for Using the IUCN Red List Categories and Criteria. Version 13*.

The IUCN Red List of Threatened Species. (2017).

IUCN/SSC. (2013). *Guidelines for reintroductions and other conservation translocations*. Retrieved from Gland, Switzerland:

Iverson, B. J. (1978). The impact of feral cats and dogs on populations of the West Indian rock iguana, *Cyclura carinata*. *Biological Conservation, 14*(1), 63-73

Iverson, B. J. (2007). Juvenile survival in the Allen Cays Rock Iguana (*Cyclura cyclura inornata*). *American Society of Ichthyologists and Herpetologists* 740- 744.

Iverson, J. B., Smith, G. R., Pasachnik, S. A., Hines, K. N., & Pieper, L. Y. N. N. E. (2016). Growth, coloration, and demography of an introduced population of the Acklins Rock Iguana (*Cyclura rileyi nuchalis)* in the Exuma Islands, The Bahamas. Herpetological Conservation and Biology, 11, 139-153.

Izurieta, A. I. (1991). *Historia natural de las iguanas terrestres (Conolophus subcristatus) de la Isla Seymour Norte, Galápagos. Unpublished thesis.* Pontificia Universidad Católica del Ecuador, Quito.

Jackson, M. H. (1993). *Galapagos, a natural history*. University of Calgary press.

Jacobson, A. P., Gerngross, P., Lemeris Jr, J. R., Schoonover, R. F., Anco, C., Breitenmoser-Würsten, C., Kamler, J. F. (2016). Leopard (*Panthera pardus*) status, distribution, and the research efforts across its range. PeerJ, 4, e1974.

Jaffe, A. L., Slater, G. J., & Alfaro, M. E. (2011). The evolution of island gigantism and body size variation in tortoises and turtles. *Biology letters*, rsbl20101084.

Jarić, I., Roberts, D. L., Gessner, J., Solow, A. R., & Courchamp, F. (2017). Science responses to IUCN Red Listing. *PeerJ, 5*

Jones, H. P., Holmes, N. D., Butchart, S. H. M., Tershy, B. R., Kappes, P. J., Corkery, I., .Croll, D. A. (2016). Invasive mammal eradication on islands results in substantial conservation gains (Vol. 113, pp. 4033-4038).

Kacoliris, F. P., Berkunsky, I., & Williams, J., D. (2009). Methods for assessing population size in sand dune lizards (*Liolaemus multimaculatus*). *Herpetologica*(2), 219.

Keedwell, R. J. (2004). *Use of population viability analysis in conservation management in New Zealand*: Wellington, N.Z. : Dept. of Conservation, c2004.

Kier, G., Kreft, H., Lee, T. M., Jetz, W., Ibisch, P. L., Nowicki, C.,Crane, P. R. (2009). A global assessment of endemism and species richness across island and mainland regions. *Proceedings of the National Academy of Sciences of the United States of America*(23), 9322.

Kimble, D. S., Tyers, D. B., Robison-Cox, J., & Sowell, B. F. (2011). Aspen recovery since wolf reintroduction on the northern yellowstone winter range. *Rangeland Ecology & Management, 64*(2), 119-130.

Knapp, C. R. (2001). Status of a translocated Cyclura iguana colony in the Bahamas (Vol. 35, pp. 239-248).

Knapp, C. R., Alvarez-Clare, S., & Perez-Heydrich, C. (2010). The influence of landscape heterogeneity and dispersal on survival of neonate insular iguanas. *Copeia*(1), 62-70.

Knapp, C. R., & Owens, A. K. (2005). Home range and habitat associations of a Bahamian iguana: implications for conservation. *Animal Conservation, 8*(3), 269-278.

Konecny, M. J. (1987). Home range and activity patterns of feral house cats in the Galápagos Islands. *Oikos*(1), 17.

Kruuk, H., & Snell, H. (1981). Prey selection by feral dogs from a population of marine iguanas (*Amblyrhynchus cristatus*). Journal of Applied Ecology, 197-204.

Kumar, K., Ortiz-Catedral, L., & Llerena, A. J. (2019). *Conservation status and demographics of the Galapagos Land Iguana (Conolophus subcristatus)* (Masters of Natural Science in Conservation Biology), Massey Univeristy, Auckland, New Zealand.

Kwiatkowski, M. A., & Sullivan, B. K. (2002). Mating system structure and population density in a polygynous lizard, Sauromalus obesus (=ater). *Behavioral Ecology*(2), 201.

Lamb, T., Jones, T. R., & Avise, J. C. (1992). Phylogeographic histories of representative herpetofauna of the southwestern US: mitochondrial DNA variation in the desert iguana (*Dipsosaurus dorsalis*) and the chuckwalla (*Sauromalus obesus*). Journal of Evolutionary Biology, 5(3), 465-480.

Lande, R. (1988). Genetics and demography in biological conservation. *Science*(4872), 1455.

Laufenberg, J. S., Clark, J. D., & Chandler, R. B. (2018). Estimating population extinction thresholds with categorical classification trees for Louisiana black bears. *Plos One, 13*(1), e0191435.

Laurie, A. (1990). Effects of the 1982-83 El Niño-southern oscillation event on Marine Iguana (*Amblyrhynchus cristatus* Bell, 1825) populations on Galapagos. *Elsevier Oceanography Series, 52*, 361-380
.

Laurie, W., & Brown, D. (1990). Population biology of marine iguanas (*Amblyrhynchus cristatus*). III. Factors affecting survival. *Journal of Animal Ecology, 59*(2), 545- 568.

Lee, D. E., & Bond, M. L. (2016). Precision, accuracy, and costs of survey methods for giraffe *Giraffa camelopardalis*. *Journal of Mammalogy, 97*(3), 940-948.

Lee, K., Huveneers, C., Gimenez, O., Peddemors, V., & Harcourt, R. (2014). To catch or to sight? A comparison of demographic parameter estimates obtained from mark-recapture and mark-resight models. *Biodiversity and Conservation, 23*(11), 2781-2800.

Lettink, M. (2012). *Herpetofauna: population estimates (using capture-mark-recapture data)*. Retrieved from Department of Conservation.

Lettink, M., Armstrong, D. P., & Pryde, M. A. (2003). *Mark-recapture analysis for monitoring threatened species : introduction and case study*: Wellington, N.Z. : Dept. of Conservation, c2003.

Letty, J., Marchandeau, S., & Aubineau, J. (2007). Problems encountered by individuals in animal translocations: lessons from field studies. *Ecoscience, 14*(4), 420-431.

Li, B., Belasen, A., Pafilis, P., Bednekoff, P., & Foufopoulos, J. (2014). Effects of feral cats on the evolution of anti-predator behaviours in island reptiles: insights from an ancient introduction. *Proceedings: Biological Sciences*(1788), 1.

Linklater, W. (2001). *Estimating Kaimanawa feral horse population size and growth*: Wellington, N.Z. : Dept. of Conservation, 2001.

Lomolino, M. V. (1985). Body size of mammals on islands: the island rule reexamined. *The American Naturalist, 125*(2), 310-316.

Lomolino, M. V. (2005). Body size evolution in insular vertebrates: generality of the island rule. *Journal of Biogeography, 32*(10), 1683-1699.

Lovegrove, T. G. (1996). Island releases of saddlebacks *Philesturnus carunculatus* in New Zealand. *Biological Conservation, 77*(2-3), 151-157.

MacLeod, A., Irisarri, I., Vences, M., & Steinfartz, S. (2016). The complete mitochondrial genomes of the Galápagos iguanas, *Amblyrhynchus cristatus* and *Conolophus subcristatus*. *Mitochondrial DNA. Part A, DNA Mapping, Sequencing, And Analysis, 27*(5), 3699-3700.

MacLeod, A., Unsworth, L., Steinfartz, S., & Trillmich, F. (2016). Mark-resight estimates confirm a critically small population size in threatened marine iguanas (*Amblyrhynchus cristatus*) on San Cristobal Island, Galapagos. *Salamandra, 52*, 58-62.

Maes, D., Isaac, N. J. B., Harrower, C. A., Collen, B., Strien, A. J., & Roy, D. B. (2015). The use of opportunistic data for IUCN Red List assessments. *Biological Journal of the Linnean Society, 115*(3), 690-706.

Malone, C. L., Knapp, C. R., & Taylor, J. F. (2003). Genetic consequences of Pleistocene fragmentation: Isolation, drift, and loss of diversity in rock iguanas (*Cyclura*). *Conservation Genetics, 4*(1), 1-15.

Malone, C. L., Wheeler, T., Taylor, J. F., & Davis, S. K. (2000). Phylogeography of the Caribbean rock iguana (*Cyclura*): implications for conservation and insights on the biogeographic history of the West Indies. *Molecular Phylogenetics and Evolution, 17*(2), 269-279.

Manuel, N., Aurelio, M., Bernie, R. T., Donlan, C. J., Dick, V., Néstor, P., . . . Jesús, A. (2004). A review of feral cat eradication on islands. *Conservation Biology*(2), 310.

Mares, M. A., Streilein, K. E., & Willig, M. R. (1981). Article navigation experimental assessment of several population estimation techniques on an introduced population of Eastern Chipmunks. *Journal of Mammalogy, 62*(2), 315-328.

Marquez, C. M., Munoz, E. A., Gentile, G., Tapia, W. H., Zabala, J. F., Naranjo, S. A., & Llerena, A. J. (2010). Estado poblacional de las iguanas terrestres (*Conolophus subcristatus, C. pallidus y C. marthae: Squamata, Iguanidae*), Islas Galápagos. *Bioletín Técnico: Serie Zoológica, 9*(6), 25-43.

McCallum, M. L. (2007). Amphibian decline or extinction? Current declines dwarf background extinction rate (Vol. 41, pp. 483-491).

McDonald, J. H. (2009). Handbook of biological statistics (Vol. 2): sparky house publishing Baltimore, MD.

McGinnis, S. M., & Brown, C. W. (1966). Thermal behavior of the green iguana, iguana iguana. *Herpetologica*(3), 189.

McKinney, M. L. (1999). High rates of extinction and threat in poorly studied taxa. *Conservation Biology*(6), 1273.

Medina, F. M., Bonnaud, E., Vidal, E., Tershy, B. R., Zavaleta, E. S., Donlan, C. J., Nogales, M. (2011). A global review of the impacts of invasive cats on island endangered vertebrates. *Global Change Biology, 17*(11), 3503-3510.

Miller, B., Reading, R. P., Ralls, K., Scott, J. M., & Estes, J. (1999). Biological and technical considerations of carnivore translocation: a review. *Animal Conservation, 2*(1), 59-68.

Milligan, S., Brown, L., Stenhouse, G., Hobson, D., & Frame, P. (2018). Factors affecting the success of grizzly bear translocations. *Journal of Wildlife Management, 82*(3), 519-530.

Miralles, A., Macleod, A. M. Y., RodrÍGuez, A., IbÁÑEz, A., JimÉNez-Categui, G., Quezada, G., . . . Steinfartz, S. (2017). Shedding light on the Imps of Darkness: an integrative taxonomic revision of the Galápagos marine iguanas (genus *Amblyrhynchus*). *Zoological Journal of the Linnean Society, 181*(3), 678-710.

Mitchell, N., C. . (1999). Effect of introduced ungulates on density, dietary preferences, home range, and physical condition of the iguana (*Cyclura pinguis*) on anegada. *Herpetologica*(1), 7.

Mitchell, N., Haeffner, R., Veer, V., Fulford-Gardener, M., Clerveaux, W., Veitch, C. R., & Mitchell, G. (2002). Cat eradication and the restoration of endangered iguanas (Cyclura carinata) on Long Cay, Caicos Bank, Turks and Caicos Islands, British West Indies. *Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group.*, 206-212.

Moore, J. A., Grant, T., Brown, D., Keall, S. N., & Nelson, N. J. (2010). Mark-recapture accurately estimates census for Tuatara, a burrowing reptile. *The Journal of Wildlife Management*, 74(4), 897-901.

Morris, R. F. (1955). Population studies on some small forest mammals in eastern canada. *Journal of Mammalogy*(1), 21.

Morrison, C., Keppel, G., Thomas, N., Rounds, I., & Harlow, P. S. (2009). Critically endangered Fijian crested iguana (*Brachylophus vitiensis*) shows habitat preference for globally threatened tropical dry forest. *Pacific Science, 63*(2), 223-242.

Morrison, C., Thomas, N., Osborne, T., Harlow, P. S., Biciloa, P., & Niukula, J. (2007). Diet and habitat preferences of the Fijian crested iguana (*Brachylophus vitiensis*) on Yadua Taba, Fiji: implications for conservation. *Australian Journal of Zoology, 55*(6), 341-350.

Mrosovsky, N. (1997). IUCN's credibility critically endangered. *Nature, 389*(6650), 436.

Mueller-Dombois, D., & Fosberg, F. R. (1998). *Vegetation of the tropical Pacific islands*: New York : Springer, c1998.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature, 403*(6772), 853-858.

Myers, R. A., Baum, J. K., Shepherd, T. D., Powers, S. P., & Peterson, C. H. (2007). Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science*(5820), 1846.

Mysterud, A. (2011). Selective harvesting of large mammals: how often does it result in directional selection? *Journal of Applied Ecology, 48*(4), 827-834.

Nogales, M., Vidal, E., Medina, F. M., Bonnaud, E., Tershy, B., Campbell, K. J., & Zavaleta, E. S. (2013). Feral Cats and Biodiversity Conservation: The Urgent Prioritization of Island Management. *BioScience, 63*(10), 804-810.

Nolet, B., & Baveco, J. (1996). Development and viability of a translocated beaver *Castor fiber* population in the Netherlands. *Biological Conservation, 75*(2), 125-137.

O'Brien, T. G., Pollock, K. H., Davidson, W. R., & Kellogg, F. E. (1985). A comparison of capture-recapture with capture-removal for quail populations. *The Journal of Wildlife Management*(4), 1062.

Palkovacs, E. P. (2003). Explaining adaptive shifts in body size on islands: a life history approach. *Oikos, 103*(1), 37-44.

Palmeirim, J. (2007). The decline of the endemic Fijian crested iguana *Brachylophus vitiensis* in the Yasawa and Mamanuca archipelagos, western Fiji. *ORYX, 41*(1), 44-50.

Palomares, F., Godoy, J. A., LÓPez- B. A. O., J. V., RodrÍGuez, A., Roques, S., Casas-Marce, M., Delibes, M. (2012). Possible extinction vortex for a population of iberian lynx on the verge of extirpation. *Conservation Biology*(4), 689.

Parent, C. E., Caccone, A., & Petren, K. (2008). Colonization and diversification of Galápagos terrestrial fauna: a phylogenetic and biogeographical synthesis. *Philosophical Transactions of the Royal Society B: Biological Sciences, 363*(1508), 3347-3361.

Parker, K. A., Ewen, J. G., Seddon, P. J., & Armstrong, D. P. (2013). Post-release monitoring of bird translocations: Why is it important and how do we do it? *Notornis, 60*(1), 85-92.

Patton, J. L., Pardiñas, U. F., & D'Elía, G. (2015). *Mammals of South America, volume 2: rodents* (Vol. 2): University of Chicago Press.

Patton, J. L., Yang, S. Y., & Myers, P. (1975). Genetic and morphologic divergence among introduced rat populations (Rattus rattus) of the Galapagos Archipelago, Ecuador. *Systematic Biology, 24*(3), 296-310.

Pech, R., & Maitland, M. (2016). Conservation of native fauna in highly invaded systems: managing mammalian predators in New Zealand. *Restoration Ecology, 24*(6), 816-820.

Pérez-Buitrago, N., Sabat, A. M., & McMillan, W. O. (2010). Spatial ecology of the endangered Mona Island iguana *Cyclura cornuta stejnegeri*: does territorial behavior regulate density? *Herpetological Monographs*, 86.

Perkins, C. B. 1930. Galapagos Islands Trip December 4th 1931 to February 27th 1932. Archive PER 5924.63 Charles Darwin Research Station Library, Puerto Ayora, Santa Cruz, Galapagos Islands, Ecuador.

Perring, M. P., Standish, R. J., Price, J. N., Craig, M. D., Erickson, T. E., Ruthrof, K. X., . . . Hobbs, R. J. (2015). Advances in restoration ecology: rising to the challenges of the coming decades. *Ecosphere, 6*(8), 1-25.

Phillips, J., A. (1995). Does cadence of *Iguana iguana* displays facilitate individual recognition? *Behavioral Ecology and Sociobiology*(5), 337.

Phillips, R. B., Cooke, B. D., Campbell, K., Carrion, V., Marquez, C., & Snell, H. L. (2005). Eradicating feral cats to protect Galapagos land iguanas: methods and strategies. *Pacific Conservation Biology, 11*, 257-267.

Phillips, R. B., Wiedenfeld, D. A., & Snell, H. L. (2012). Current status of alien vertebrates in the Galápagos Islands: invasion history, distribution, and potential impacts. *Biological Invasions*, 14(2), 461-480.

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., Sexton, J. O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection (Vol. 344, pp. 987-+).

Pimm, S. L., Russell, G. J., Gittleman, J. L., & Brooks, T. M. (1995). The future of biodiversity. *Science*(5222), 347.

Pincheira-Donoso, D., Bauer, A. M., Meiri, S., & Uetz, P. (2013). Global taxonomic diversity of living reptiles. *Plos One, 8*(3), 1-10.

Pollock, K. H., Nichols, J. D., Simons, T. R., Farnsworth, G. L., Bailey, L. L., & Sauer, J. R. (2002). Large scale wildlife monitoring studies: statistical methods for design and analysis (Vol. 13, pp. 105-119).

Pregill, G. K., & Steadman, D. W. (2004). South pacific iguanas: human impacts and a new species (Vol. 38, pp. 15-21).

Proudlove, G. S. (2004). Conservation assessments, threats, and the iucn red list categories and criteria. *Journal of Biogeography*(7), 1201.

Pyron, R. A., Burbrink, F. T., & Wiens, J. J. (2013). A phylogeny and revised classification of Squamata, including 4161 species of lizards and snakes. *BMC Evolutionary Biology, 13*(1), 1-53.

Rassmann, K., Gliddon, C., Tautz, D., & Trillmich, F. (1997). The microevolution of the Galapagos marine iguana *Amblyrhynchus cristatus* assessed by nuclear and mitochondrial genetic analyses. *Molecular Ecology, 6*(5), 437-452.

Rassmann, K., Trillmich, F., & Tautz, D. (1997). Hybridization between the Galapagos land and marine iguana (*Conolophus subcristatus* and *Amblyrhynchus cristatus*) on Plaza Sur. *Journal of Zoology, 242*(4), 729-739.

Reed, D. H., O'Grady, J. J., Brook, B. W., Ballou, J. D., & Frankham, R. (2003). Estimates of minimum viable population sizes for vertebrates and factors influencing those estimates. *Biological Conservation, 113*(1), 23-34.

Rees, S. G., Goodenough, A. E., Hart, A. G., & Stafford, R. (2011). Testing the effectiveness of capture mark recapture population estimation techniques using a computer simulation with known population size. *Ecological Modelling, 222*(17), 3291-3294.

Reichert, B. E., Fletcher, R. J., Kitchens, W. M., & Kendall, W. L. (2016). Spatio-temporal variation in age structure and abundance of the endangered snail kite: Pooling across regions masks a declining and aging population. *Plos One, 11*(9).

Reynolds, R. (1982). Experimental repatriation of captive-reared land iguanas (*Conolophus subcristatus*) at Cartago Bay, Isabela. *Noticias De Galapagos.* 34, 17-18

Ripple, W. J., & Beschta, R. L. (2003). Wolf reintroduction, predation risk, and cottonwood recovery in Yellowstone National Park. *Forest Ecology and Management, 184*(1-3), 299-313.

Ripple, W. J., & Beschta, R. L. (2004). Wolves, elk, willows, and trophic cascades in the upper Gallatin Range of Southwestern Montana, USA. *Forest Ecology and Management, 200*(1-3), 161-181.

Ripple, W. J., & Larsen, E. J. (2000). Historic aspen recruitment, elk, and wolves in northern Yellowstone National Park, USA. *Biological Conservation, 95*(3), 361-370.

Rockstrom, J., Steffen, W., Noone, K., Person, A., Chapin, S. F., Lambin, E. F., Foley, J. A. (2009). A safe operating space for humanity. *Nature*(7263), 472.

Rodda, G. H., & Campbell, E. W. (2002). Distance sampling of forest snakes and lizards. *Herpetological Review, 33*(4), 271-274.

Rodrigues, A. S. L., Pilgrim, J. D., Lamoreux, J. F., Hoffmann, M., & Brooks, T. M. (2006). The value of the IUCN Red List for conservation. *Trends In Ecology & Evolution, 21*, 71-76.

Ruiz de Infante Anton, J., Travecchia, G., Igual, J. M., & Rotger, A. (2013). Estimating lizard population density: an empirical comparison between line-transect and capture–recapture methods. *Wildlide Research, 40*(7), 552-560.

Russell, J. C., Innes, J. G., Brown, P. H., & Byrom, A. E. (2015). Predator-free New Zealand: conservation country. *BioScience, 65*(5), 520-525.

Santamaria, L., Fernandez Mendez, P., Santamaria, L., & Mendez, P. F. (2012). Evolution in biodiversity policy - current gaps and future needs. *Evolutionary Applications, 5*(2), 202-218.

Schatz, H. (1998). Review Oribatid mites of the Galápagos Islands–faunistics, ecology and speciation. Experimental & applied acarology, 22(7), 373-409.

Schlacher, T. A., Lucrezi, S., Peterson, C. H., Connolly, R. M., Olds, A. D., Althaus, F., Leon, J. X. (2016). Estimating animal populations and body sizes from burrows: Marine ecologists have their heads buried in the sand. *Journal of sea research, 112*, 55-64.

Schmidt, M. R., & Wei, W. (2006). Loss of agro-biodiversity, uncertainty, and perceived control: a comparative risk perception study in Austria and China. *Risk Analysis: An International Journal, 26*(2), 455-470.

Schnabel, Z. E. (1938). The estimation of total fish population of a lake. *The American Mathematical Monthly*(6), 348.

Schulte, J. A., Valladares, J. P., & Larson, A. (2003). Phylogenetic relationships within iguanidae inferred using molecular and morphological data and a phylogenetic taxonomy of iguanian lizards. *Herpetologica*(3), 399.

Science and the Endangered Species Act. (1995). Washington, DC : National Academy Press, 1995.

Seber, G. A. F. (1970). The effects of trap response on tag recapture estimates. *Biometrics*(1), 13.

Seber, G. A. F. (1982). *The estimation of animal abundance and related parameters*: London : Charles Griffin, 1982

Seddon, P. (2010). From reintroduction to assisted colonization: moving along the conservation translocation spectrum. *Restoration Ecology, 18*(6), 796-802.

Seddon, P., Seddon, P. J., Griffiths, C. J., Soorae, P. S., & Armstrong, D. P. (2014). Reversing defaunation: Restoring species in a changing world. *Science, 345*(6195), 406-412.

Seddon, P. J., Armstrong, D. P., & Maloney, R. F. (2007). Developing the science of reintroduction biology. *Conservation Biology*(2), 303.

Sementelli, A., Smith, H. T., Meshaka Jr, W. E., & Engeman, R. M. (2008). Just green iguanas? The associated costs and policy implications of exotic invasive wildlife in South Florida. *Public works management & policy, 12*(4), 599-606.

Shine, R., & Harlow, P. S. (1998). Ecological divergence among sympatric colour morphs in blood pythons, *Python brongersmai*. *Oecologia, 116*(1-2), 113-119.

Sites Jr, J. W., Davis, S. K., Guerra, T., Iverson, J. B., & Snell, H. L. (1996). Character congruence and phylogenetic signal in molecular and morphological data sets: A case study in the living Iguanas (*squamata, Iguanidae*). *Molecular Biology and Evolution, 13*(8), 1087-1105.

Smith, F. D. M., May, R. M., Pellew, R., Johnson, T. H., & Walter, K. R. (1993). How much do we know about the current extinction rate? *Trends in Ecology and Evolution, 8*(10), 375-378.

Smolensky, N. L., & Fitzgerald, L. A. (2010). Distance sampling underestimates population densities of dune-dwelling lizards. *Journal of Herpetology, 44*(3), 372-381.

Snell, H. 1994. The history of the Cerro Dragon, Santa Cruz, Land Iguana Conservation Project. Archive SNE 397.95 Charles Darwin Research Station Library, Puerto Ayora, Santa Cruz, Galapagos Islands, Ecuador.

Snell, H. H., Snell, M. H., & Tracy, C. R. (1984). Variation among populations of Galapagos land iguanas (*Conolophus*): contrasts of phylogeny and ecology. *Biological Journal of the Linnean Sociey*(21), 185-207.

Snell, H. L., & Tracy, R. C. (1985). Behavioral and morphological adaptations by Galapagos land iguanas (*Conolophus subcristatus*) to water and energy requirements of eggs and neonates. *American Zoologist, 25*(4), 1009-1018.

Spatz, D. R., Zilliacus, K. M., Holmes, N. D., Butchart, S. H. M., Genovesi, P., Ceballos, G., Croll, D. A. (2017). Globally threatened vertebrates on islands with invasive species. *Science Advances, 2*(10).

Steadman, D. W., Stafford, T. W., Donahue, D. J., & Jull, A. (1991). Chronology of Holocene vertebrate extinction in the Galápagos Islands. Quaternary research, 36(1), 126-133.

Steadman, D. W., & Zousmer, S. (1988). *Galapagos: discovery on Darwin's islands*: Smithsonian Institution.

Steindachner, F. (1876). *Die Schlangen und Eidechsen der Galapagos-Inseln*: Wien :K.K. Zoologisch-botanischen gesellschaft.

Stringer, A. P., & Gaywood, M. J. (2016). The impacts of beavers *Castor spp.* on biodiversity and the ecological basis for their reintroduction to Scotland, UK. *Mammal Review, 46*(4), 270-283.

Sutherland, W. J. (2006). *Ecological census techniques : a handbook*: Cambridge : Cambridge University Press, 2006, 2nd ed.

Switak, K. H. (1998). Galapagos land iguanas: surviving in peril. *Iguana Times, 7*(1), 3-10.

Syfert, M. M., Joppa, L., Smith, M. J., Coomes, D. A., Bachman, S. P., & Brummitt, N. A. (2014). Using species distribution models to inform IUCN Red List assessments. *Biological Conservation, 177*, 174-184.

Szucs, M., Melbourne, B. A., Tuff, T., & Hufbauer, R. A. (2014). The roles of demography and genetics in the early stages of colonization (Vol. 281).

Tarszisz, E., Dickman, C. R., & Munn, A. J. (2014). Physiology in conservation translocations (Vol. 2).

Taylor, B. W., Flecker, A. S., & O. Hall, R. J. (2006). Loss of a harvested fish species disrupts carbon flow in a diverse tropical river. *Science*(5788), 833.

Tenan, S., Rotger Vallespir, A., Igual, J. M., Moya, Ó., Royle, J. A., & Tavecchia, G. (2013). Population abundance, size structure and sex-ratio in an insular lizard. *Ecological Modelling, 267*, 39-47.

Tershy, B., Newton, M. K., Spatz, R. D., Swinnerton, J. K., Iverson, B. J., Fisher, N. R., Croll, A. D. (2016). The biogeography of threatened insular iguanas and opportunities for invasive vertebrate management. *Herpetological Conservation & Biology, 11*, 222-236.

Tershy, B. R., Kuo-Wei, S., Newton, K. M., Croll, D. A., & Holmes, N. D. (2015). The importance of islands for the protection of biological and linguistic diversity. *BioScience, 65*(6), 592-597.

Thomas, J. A., Telfer, M. G., Roy, D. B., Preston, C. D., Greenwood, J. J. D., Asher, J., Lawton, J. H. (2004). comparative losses of british butterflies, birds, and plants and the global extinction crisis. *Science*(5665), 1879.

Thomas, L., Buckland, S., Rexstad, E. A., Bishop, J. R. B., Marques, T. A., Laake, J. L., Burnham, K. P. (2010). Distance software: design and analysis of distance sampling surveys for estimating population size. *Journal of Applied Ecology, 47*(1), 5-14.

Tinoco, B. A., Astudillo, P. X., Latta, S. C., & Graham, C. H. (2009). Distribution, ecology and conservation of an endangered Andean hummingbird: the Violet-throated Metaltail (*Metallura baroni*). *Bird Conservation International, 19*(1), 63-76.

Titley, M. A., Snaddon, J. L., & Turner, E. C. (2017). Scientific research on animal biodiversity is systematically biased towards vertebrates and temperate regions. *Plos One, 12*(12), e0189577.

Tole, L. (2002). Habitat loss and anthropogenic disturbance in Jamaica's Hellshire Hills area (Vol. 11, pp. 575-598).

Towns, D. R., Daugherty, C. H., & Cree, A. (2001). Raising the prospects for a forgotten fauna: a review of 10 years of conservation effort for New Zealand reptiles. *Biological Conservation, 99*(1), 3-16.

Towns, D. R., Miller, K. A., Nelson, N. J., & Chapple, D. G. (2016). Can translocations to islands reduce extinction risk for reptiles? case studies from New Zealand. *Biological Conservation, 204*, 120-127.

Townsend Peterson, A., Papeş, M., & Eaton, M. (2007). Transferability and model evaluation in ecological niche modeling: a comparison of GARP and Maxent. *Ecography, 30*(4), 550-560.

Traveset, A. (1990). *Ctenosaura similis* Gray (*Iguanidae*) as a seed disperser in a central american deciduous forest. *The American Midland Naturalist*(2), 402.

Traveset, A., Nogales, M., Vargas, P., Rumeu, B., Heleno, R., Olesen, J. M., & Jaramillo, P. (2016). Galápagos land iguana (*Conolophus subcristatus*) as a seed disperser. *Integrative Zoology, 11*(3), 207-213.

Tzika, A. C., Rosa, S. F. P., Fabiani, A., Snell, H. L., Snell, H. M., Marquez, C., Milinkovitch, M. C. (2008). Population genetics of Galapagos land iguana (genus *Conolophus*) remnant populations. *Molecular Ecology, 17*(23), 4943-4952.

Valiente-Banuet, A., Aizen, M. A., Alcántara, J. M., Arroyo, J., Cocucci, A., Galetti, M., Zamora, R. (2015). Beyond species loss: the extinction of ecological interactions in a changing world. *Functional Ecology, 29*(3), 299-307.

van den Burg, M. P., Meirmans, P. G., van Wagensveld, T. P., Kluskens, B., Madden, H., Welch, M. E., & Breeuwer, J. A. J. (2018). The Lesser Antillean Iguana (*Iguana delicatissima*) on St. Eustatius: genetically depauperate and threatened by ongoing hybridization. *Journal of Heredity, 109*(4), 426.

Van Horne, B., Schooley, R. L., Knick, S. T., Olson, G. S., & Burnham, K. P. (1997). Use of burrow entrances to indicate densities of Townsend's ground squirrels. *The Journal of Wildlife Management*, 92-101.

Van Katwyk, K. E. (2014). *Empirical validation of closed population abundance estimates and spatially explicit density estimates using a censused population of North American red squirrels.* (Masters of Science), University of Alberta, Edmonton, Alberta.

van Marken Lichtenbelt, W. D. (1993). Optimal foraging of a herbivorous lizard, the green iguana in a seasonal environment. *Oecologia*(2), 246.

Vasconez, F. J., Ramón, P., Hernandez, S., Hidalgo, S., Bernard, B., Ruiz, M., Ruiz, G. (2018). The different characteristics of the recent eruptions of Fernandina and Sierra Negra volcanoes (Galápagos, Ecuador). *Volcanica, 1*(2), 127-133.

Vié, J.-C., Hilton-Taylor, C., Pollock, C., Ragle, J., Smart, J., Stuart, S. N., & Tong, R. (2009). The IUCN Red List: a key conservation tool *Wildlife in a changing world–An analysis of the 2008 IUCN Red List of Threatened Species* (pp. 1).

Wanger, T. C., Motzke, I., Furrer, S. C., Brook, B. W., & Gruber, B. (2009). How to monitor elusive lizards: comparison of capture–recapture methods on giant day geckos (*Gekkonidae, Phelsuma madagascariensis grandis*) in the Masoala rainforest exhibit, Zurich Zoo. *Ecological Research, 24*(2), 345-353.

Welch, M., Colosimo, G., Pasachnik, S., Malone, C., Hilton, J., Long, J., Gerber, G. (2017). Molecular variation and population structure in critically endangered Turks and Caicos Rock Iguanas: identifying intraspecific conservation units and revising subspecific taxonomy. *Conservation Genetics, 18*(2), 479.

Werner, D. 1976. The Galapagos land Iguana (*Conolophus*): Natural history and conservation survey. National Geographic Society Research Reports Vol 17. Archive WER 597.95 Charles Darwin Research Station Library, Puerto Ayora, Santa Cruz, Galapagos Islands, Ecuador.

Werner, D. I. (1982). Social organization and ecology of land iguanas, *Conolophus subcristatus*, on isla Fernandina, Galapagos. Park Ridge, USA: Neyes Publications.

Werner, D. I. (1983). Reproduction in the iguana *Conolophus subcristatus* on Fernandina island, galapagos: clutch size and migration costs. *The American Naturalist,* 757-775.

White, A., & Pyke, G. (2008). Frogs on the hop: translocations of Green and Golden Bell Frogs Litoria aurea in Greater Sydney. *Australian Zoologist, 34*(3), 249-260.

Wiedenfeld, D. A., & Jiménez-Uzcátegui, G. (2008). Critical problems for bird conservation in the Galápagos Islands. Cotinga, 29, 22-27.

Wiens, J., J., & Hollingsworth, B., D. (2000). War of the iguanas: conflicting molecular and morphological phylogenies and long-branch attraction in iguanid lizards. *Systematic Biology*(1), 143.

Wikelski, M. (2005). Evolution of body size in Galapagos marine iguanas. *Proceedings. Biological Sciences, 272*(1576), 1985-1993.

Williams, B. K., Conroy, M. J., & Nichols, J. D. (2002). *Analysis and management of animal populations : modeling, estimation, and decision making*: London : APUK, 2002.

Wilson, B., Hammond, P. S., & Thompson, P. M. (1999). Estimating size and assessing trends in a coastal bottlenose dolphin population. *Ecological Applications, 9*(1), 288.

Wilson, B. S., Robinson, O. F., & Vogel, P. (2004). Status of the jamaican iguana (*Cyclura collei*): Assessing 15 years of conservation effort. *Iguana, 11*, 225-231.

Wood, J. R., Wilmshurst, J. M., Alcover, J. A., Bover, P., Blackburn, T. M., Duncan, R. P., Rando, J. C. (2017). Island extinctions: Processes, patterns, and potential for ecosystem restoration. *Environmental Conservation, 44*(4), 348-358.

Woram, J.M. 1991. Who killed the iguanas? *Noticias de Galapagos*. 50, 12-17

Worthy, T. H., Tennyson, A. J., Archer, M., Musser, A. M., Hand, S. J., Jones, C., Beck, R. M. (2006). Miocene mammal reveals a Mesozoic ghost lineage on insular New Zealand, southwest Pacific. Proceedings of the National Academy of Sciences, 103(51), 19419-19423.

WWF. (2016). *Stranger species*. Retrieved from WWF

Young, T. P. (2000). Restoration ecology and conservation biology. *Biological Conservation, 92*, 73-83.

Zaher, H., Yánez-Muñoz, M. H., Rodrigues, M. T., Graboski, R., Machado, F. A., Altamirano-Benavides, M., Grazziotin, F. G. (2018). Origin and hidden diversity within the poorly known Galápagos snake radiation (Serpentes: Dipsadidae). *Systematics and Biodiversity*, 1-29.

Zarza, E., Reynoso, V. H., & Emerson, B. C. (2011). Discordant patterns of geographic variation between mitochondrial and microsatellite markers in the Mexican black iguana (*Ctenosaura pectinata*) in a contact zone. *Journal of Biogeography*(7), 1394.

Zenboudji, S., Cheylan, M., Arnal, V., Bertolero, A., Leblois, R., Astruc, G.,Montgelard, C. (2016). Conservation of the endangered Mediterranean tortoise Testudo hermanni hermanni: The contribution of population genetics and historical demography. *Biological Conservation, 195*, 279-291.

APPENDIX

APPENDIX A

CATALOGUE OF GALAPAGOS LAND IGUANA PHOTOGRAPHS FROM NORTH SEYMOUR AND FERNANDINA ISLAND

Land iguanas caught on North Seymour Island

Males



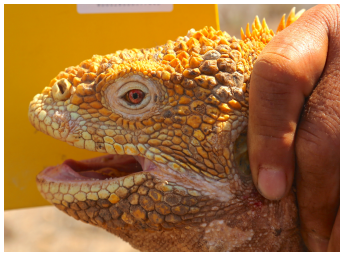
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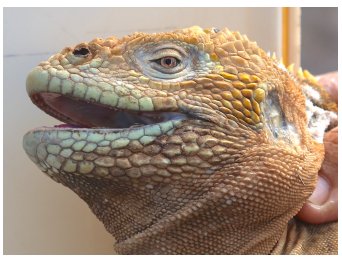
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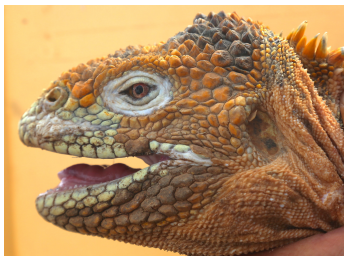
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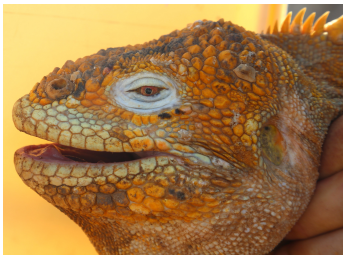
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Females



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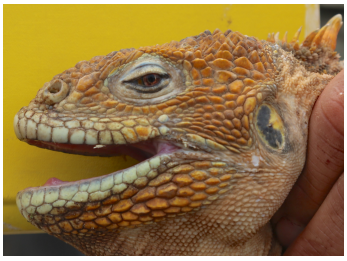
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Juveniles and subadults



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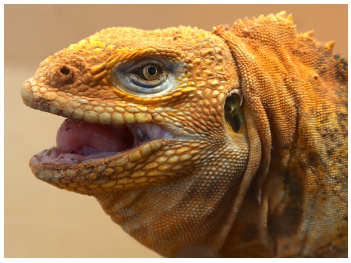
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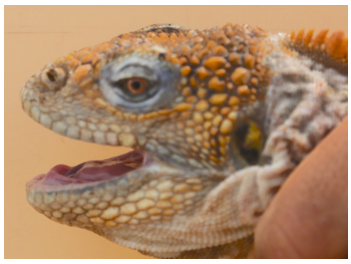
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Land iguanas caught on Fernandina Island

Males



071-806-066



071-835-002



106-562-048



106-564-111



106-565-331



106-565-560



106-566-808



106-572-631



106-593-315



900026000577479



900026000577485



900026000577488



900026000577500



900026000577501



900026000577510



900026000577512



900026000577522



900026000577523



900026000577528



900026000577532



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900026000577536



900026000577540



900026000577548



900026000577551



900026000577605



900026000577612



900026000577614



900026000577615



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900026000577751



900026000577752



900026000577755



900026000577757



900026000577758

Females



900026000577549



900026000577641



900026000577646



900026000577660



900026000577666



900026000577667



900026000577678



900026000577681



900026000577700



900026000577705



900026000577760

Juveniles and subadults



007D2C417



900026000577484

APPENDIX B

RESEARCH PERMIT

DIRECCIÓN DEL PARQUE NACIONAL GALÁPAGOS
DIRECCIÓN DE GESTIÓN AMBIENTAL
PERMISO DE INVESTIGACIÓN CIENTÍFICA: **N° PC-96-18**

Versión: 01 Fecha: 09/01/17

Título del Proyecto: Proyecto de translocación de la iguana terrestre de Galápagos (*Conolophus subcristatus*) a la isla Santiago como estrategia para contribuir a la restauración ecológica de la especie y el ecosistema en la isla.	
Nombre del Aplicante: Danny Rueda, Luis Ortiz- Catedral	**Contraparte Institucional:** N.A.
Dirección actual completa: Oficinas del Parque Nacional Galápagos, Pto Ayora Santa Cruz. Email: drueda@galapagos.gob.ec	
Otros participantes en el Proyecto: Christian Sevilla (DPNG), Wilson Cabrera (DPNG), Andrea Loyola (DPNG), Víctor Carrión, Karl Campbell (Island Conservation), Gabriele Gentile (U. Tor Vergata), Kirtana Kumar (Massey University, Nueva Zelanda).	
Clasificación del Proyecto: Conservación	**Requiere Contrato Marco de Acceso a Recursos Genéticos:** NO
Se requiere colectar muestras: SI	**Factura:** N.A.
Duración del Permiso de Investigación: Del 04 de diciembre del 2018 al 31 de diciembre del 2019	
Islas: Seymour Norte, Santa Cruz, Santiago.	**Sitios: Seymour Norte**, Centro de Crianza DPNG **(Santa Cruz)**, Espumilla, Bucanero, Puerto Nuevo, Puerto Rea, y Cerro Colorado arriba de la Bomba, Zona Árida, Zona de Transición **(Santiago)**.
Instituciones auspiciantes: Dirección del Parque Nacional Galápagos, Island Conservation, Massey University, Universidad Tor Vergata, Galapagos Conservancy	

Condiciones de cumplimiento obligatorio:

1. Los investigadores deberán cumplir estrictamente con todas las regulaciones establecidas en el Manual de Procedimientos para Científicos Visitantes y Protocolos para viajes de Campo y Campamentos en las Islas Galápagos.

2. Únicamente el equipo de investigadores incluido en el presente permiso está autorizado para participar durante el desarrollo del proyecto, tanto en la fase de campo, análisis de muestras, tabulación de datos y desarrollo de publicaciones. En caso de requerir la participación de investigadores adicionales, los responsables del proyecto deberán solicitar la autorización correspondiente por escrito a esta Dirección, presentado las justificaciones necesarias.

3. Previo a la salida de campo, el investigador principal deberá presentar a la DPNG el AVISO DE VIAJE DE CAMPO correspondiente, con mínimo 72 horas de anticipación y de acuerdo al formato establecido; y coordinar permanentemente con el Responsable del proceso de Conservación y restauración de Ecosistemas Insulares de la DPNG en Santa Cruz, a fin de sistematizar todas las actividades que se ejecuten dentro del proyecto.

4. Para las salidas de campo del proyecto, los investigadores deberán contar con una copia legible del respectivo Permiso de Investigación y el Aviso de Viaje de Campo firmado por los técnicos de la DPNG.

5. Los sitios e islas autorizados serán exclusivamente: **Seymour Norte**, Centro de Crianza DPNG **(Santa Cruz)**, Espumilla, Bucanero, Puerto Nuevo, Puerto Rea, y Cerro Colorado arriba de la Bomba, Zona Árida, Zona de Transición **(Santiago)**.

6. **Se autoriza** la captura y manipulación de un máximo de 4000 iguanas amarillas (*C. subcristatus*), de la isla Seymour Norte con la finalidad de ser puestas en cuarentena en instalaciones de la DPNG isla Santa Cruz, y su posterior translocación a la isla Santiago.

7. **Se autoriza** el monitoreo post-liberación (*flush transects Captura-Recaptura y Radiotelemetría)* de las iguanas amarillas (*C. subcristatus*) translocadas a la isla Santiago con la finalidad de evaluar su estado luego de la translocación desde Seymour norte.

8. **Se autoriza** el marcaje mediante la implantación de microchip (Pit Tag) el cual será colocado subcutáneamente en el total de las iguanas amarillas (*C. subcristatus*) capturadas en la isla Seymour Norte con la finalidad de facilitar su identificación.

9. **Se autoriza** la colocación de 30 transmisores (uno por individuo) VHF modelo *AL-2 Holohil Systems*, los cuales serán fijados a la cadera de los individuos de iguana amarilla (*C. subcristatus*) mediante el uso de gel de Cyanocilato.

10. **Se autoriza** el uso de Permetrina como agente de control de parásitos externos como ácaros o garrapatas mediante su aplicación sobre el cuerpo de las iguanas amarillas (*C. subcristatus*) capturadas, este compuesto deberá ser en una dilución al 0,5%.

11. **Se autoriza** la colección de muestras de un máximo de 7500 muestras de sangre (1ml c/u) de iguanas amarillas (*C. subcristatus*), para determinar nivel de Hematocrito, además estas serán subdivididas de la siguiente forma: 500 muestras de plasma sanguíneo para análisis hormonales; 2 frotis sanguíneos de cada muestra (15000 placas) para conteo diferencial de células y presencia de hemoparásitos.

12. **Se autoriza** la colección de un máximo de 40 muestras de uñas de iguanas amarillas *C. subcristatus*, con la finalidad de determinar la dieta de las mismas mediante la aplicación de análisis de isotopos estables.

13. **Se autoriza** la colección del total de las heces producidas por las iguanas amarinas (*C. subcristatus*) durante su aislamiento y cuarentena en las instalaciones de la DPNG con la finalidad de evaluar la presencia de semillas restantes en el tracto digestivo de las iguanas.

14. **Se autoriza** la realización de necropsias a los cadáveres de iguana amarilla (*C. subcristatus*) que pudieran resultar durante el periodo de aislamiento y cuarentena en las instalaciones de la DPNG en la isla Santa Cruz.

15. Posterior al cumplimiento de los procedimientos establecidos por la DPNG, los investigadores podrán movilizar las muestras autorizadas para los análisis correspondientes en los laboratorios del Veterinary Diagnostic Laboratory - Iowa State University, Ames, IA, Estados Unidos el Edward's Lab, University of California, La Merced, CA, Estados Unidos, el Conservation Metrics Inc, Santa Cruz, CA, Estados Unidos y la Universidad Tor Vergata, Roma, Italia.

16. Las muestras colectadas dentro del marco legal del presente permiso de investigación NO podrán ser utilizadas en actividades de **BIOPROSPECCIÓN, NI ACCESO AL RECURSO GENÉTICO**; y estas únicamente podrán ser utilizadas bajo las líneas de estudio autorizadas por el Ministerio del Ambiente del Ecuador a través de Dirección del Parque Nacional Galápagos.

17. El desarrollo de actividades de investigación científica dentro de las Áreas Protegidas como el Parque Nacional y la Reserva Marina de Galápagos es un privilegio concedido por el Gobierno Ecuatoriano a través de la DPNG, por lo que los responsables del proyecto deberán citar el número de Permiso de Investigación Científica otorgada por la DPNG, e incluir los reconocimientos y/o agradecimientos en las publicaciones científicas, Tesis o informes técnicos científicos que se emita como producto generado en base al presente permiso de investigación.

18. El análisis de los datos y los avances de la investigación deberán estar disponibles permanentemente para los técnicos de la DPNG, existiendo el compromiso de usarlos únicamente para acciones de manejo y no publicarlos sin el consentimiento de los investigadores principales del proyecto.

19. Previa a publicaciones científicas como resultados del proyecto, se deberá remitir el documento final ya aceptado para publicación a la DPNG bajo la denominación de embargo, con la finalidad de conocer y preparar el boletín correspondiente con dicha información, existiendo el compromiso como DPNG de no realizar ninguna difusión hasta contar con la publicación científica oficial.

20. Una vez concluido el análisis de las muestras, estas, las bibliotecas de datos y/o cualquier material resultante deberán ser devueltos a la DPNG. Estas deben ser preservadas, curadas y depositadas correctamente en las colecciones de referencia en Galápagos, de lo contrario, se deberán sufragar los gastos que demanden la preparación del material para su ingreso a la colección correspondiente.

21. Previo a la renovación del permiso de investigación, el científico responsable del proyecto o su contraparte institucional de investigación deberán entregar a la DPNG lo siguiente:

    - Una copia digital (Tabla de Excel 97-2003) de todos los datos obtenidos durante la ejecución del proyecto.
    - Un INFORME TECNICO DE CAMPO por cada Aviso De Viaje De Campo presentado a la DPNG, para la ejecución del proyecto.
    - Dos conferencias para los guardaparques de la DPNG, guías naturalistas y otras personas interesadas sobre los avances del proyecto. Para establecer el cronograma se deberá coordinar con el Responsable de Investigación Aplicada de la DPNG.
    - Un INFORME DE AVANCES del proyecto con todos los detalles de la investigación y sus recomendaciones técnicas aplicables al manejo de las Áreas Protegidas.
    - Artículos y publicaciones resultantes de la siguiente manera: en el caso de publicaciones únicamente en formato digital, si son libros o tesis, 5 impresas y una digital, en el caso de la versión digital lo harán en dispositivos magnéticos adecuadamente identificados.

DIRECCIÓN DEL PARQUE NACIONAL GALÁPAGOS
DIRECCIÓN DE GESTIÓN AMBIENTAL
PERMISO DE INVESTIGACIÓN CIENTÍFICA: **N° PC-96-18**

Versión: 01 Fecha: 09/01/17

22. Los datos que se desprendan de esta investigación, no podrán ser utilizados para estudios posteriores sin la previa autorización del Ministerio del Ambiente, a través de la Dirección del Parque Nacional Galápagos.

23. Del incumplimiento de las obligaciones dispuestas anteriormente se responsabiliza a Danny Rueda (DPNG) y Paula Castaño (Island Conservation) Responsables del proyecto. Por lo tanto el incumplimiento de cualquiera de estas condiciones así como el uso indebido de este documento, serán sancionados conforme al Código Orgánico Ambiental y dependiendo de la infracción podría conllevar a suspensión inmediata de la investigación.

Valoración Técnica: Carlos Vera	**Categoría:** COLABORADOR DPNG 2018

Considerando que el Título II, numeral 2.2, literal V del Estatuto Orgánico de Gestión por Procesos de la Dirección del Parque Nacional Galápagos publicado en la Edición Especial N° 349 del Registro Oficial publicado el martes 16 de octubre del 2012, establece entre las atribuciones y responsabilidades del Director de Gestión Ambiental "Administrar y organizar las actividades de investigación que se desarrollen en las áreas protegidas de Galápagos, en coordinación con el proceso de investigación", además que mediante Resolución N° 71 del 14 de Diciembre del 2012, el Director del Parque Nacional Galápagos delegó al Director de Gestión Ambiental, para que en su nombre y representación tramite y suscriba los actos relacionados con el desarrollo de proyectos de investigación científica en las áreas protegidas de Galápagos; en uso de la facultad delegada y de conformidad a lo señalado en el Título Cuarto, capítulo I del Estatuto Administrativo de la Dirección del Parque Nacional Galápagos, otorga el presente PERMISO DE INVESTIGACIÓN.

Reporte de Avances o Final: 15 de junio 2019 **Entrega de Propuesta para Renovación:** 01 de diciembre 2019	**FIRMA DEL INVESTIGADOR PRINCIPAL**
Fecha de emisión: 28 de diciembre de 2018	**Director de Gestión Ambiental** **DIRECCIÓN DEL PARQUE NACIONAL GALÁPAGOS** **Sello PNG**

Parque Nacional Galápagos Ecuador

APPENDIX C

FIELD REPORTS AND COMMUNICATION WITH THE GALAPAGOS NATIONAL PARK

DIRECCION DEL PARQUE NACIONAL GALAPAGOS

INFORME DE CAMPO

PROCESO: Conservación y Restauración de Ecosistemas Insulares

SUBPROCESO: Conservación de Especies Nativas

ACTIVIDAD: Monitoreo del estado poblacional de Iguanas Terrestres y colecta de muestras de Culebras en la Isla Seymour Norte.

ISLA / SITIOS: Seymour Norte

PERSONAL: Danny Rueda, Wilson Cabrera, Andrea Loyola, Marcelo Gavilanes, Luis Ortiz-Catedral, Kirtana Kumar y Enzo Reyes

PERIODO: 11 Agosto y 14 de Agosto 2017

FECHA DE PRESENTACION: 16 Agosto 2017

TRANSPORTE: Marítimo. Sea Ranger 3.

1.- INTRODUCCIÓN

Las islas Galápagos cuenta con dos áreas muy importantes: la primera declarada Parque Nacional Galápagos (PNG) que incluye el 97 % de la superficie total de las islas y la segunda declarada Reserva Marina de Galápagos (RMG) que abarca una extensión de 135.000 km² aproximadamente dentro una franja de 40 millas náuticas, medidas a partir de la línea base de las islas extremas del archipiélago, las cuales están protegidas bajo un marco legal establecido por el gobierno Ecuatoriano.

En el presente reporte se presentan los datos de colecta de muestras de tejido de cuelbras de Seymour Norte y un programa piloto de monitoreo de Iguanas terrestres en el mismo Islote. Estas actividades fueron desarrolladas en coordinación con personal de la Universidad de Massey de Nueva Zelandia, liderado por el Dr. Luis Ortiz-Catedral y personal de la Dirección General del Parque Nacional Galápagos representado por los Sres. Magister Danny Rueda y Wilson Cabrera.

2.- OBJETIVOS

1. Colecta de muestras de culebras de Galápagos en la isla Seymour Norte.
2. Monitoreo de Iguanas terrestres.

3.- MATERIALES

-Libreta de campo
-Cámara digital
-GPS
-Lector de microchips
-Pesola

-Cinta métrica.
-Binoculares
-Tubos plásticos y soluciones para colecta de tejido (1-2 mm cola de culebras)

4.- METODOLOGIA

Sitios de monitoreo

Para la captura de culebras, estas se capturaron entre las 7:30 y 17:00. Una vez capturadas manualmente las culebras fueron puestas en fundas de tela de algodón y procesadas en 25-30 minutos en promedio. Cada culebra fue medida, pesada y fotografiada. Las muestras colectadas fueron preservadas en Rna Later y en Alcohol al 97%.

Las Iguanas fueron capturadas por el Sr. Marcelo Gavilanes y medidas de acuerdo a los protocolos de campo estándar, en practica por la Dirección General del Parque Nacional Galápagos.

5.- DESARROLLO DIARIO DE ACTIVIDADES

Viernes 11 de Agosto del 2017

Salida de Puerto Ayora a las 5:45 am en la fibra Ranger 3. Llegada al Canal de Itabaca a las 6:30 para recoger personal. Llegada a Seymour Norte a las 7:00 am. Poco después de la llegada al islote, Se procedió a capturar un cachorro de lobo marino que tenia un plástico alrededor del cuello. Después de que esto se removió se libero al cachorro y se comenzó con la captura de culebras a las 7:30 am. La colecta de campo comenzó cerca al sendero turístico pero a las 11 am el personal se traslado al área conocida como La Plataforma. En esta se trabajo hasta las 17:30 horas. En total fueron capturadas 15 culebras en su mayoría adultas (ver Tabla 1).

Lunes 14 de Agosto del 2017

Salida de Puerto Ayora en fibra Sea Ranger 3 a las 5:30 am. Llegada al Canal de Itabaca a las 6:45 am para recoger personal y trasladar a Seymour Norte. Se Desembarco directamente en el área de La Plataforma a las 7:15 am. La captura de Iguanas comenzó a las 7:20 y se extendió hasta las 16:00 hrs. En total se capturaron 33 Iguanas, de las cuales 31 fueron nuevas y 2 recapturas. Además se completaron 6 transectos de entre 100 y 200 m de largo usando la metodología DISTANCE para estimar la densidad de Iguanas por hectárea en el área de La Plataforma. En la Tabla 1 se presentan los datos de las iguanas capturadas. En la Figura 1 se presentan las localidades de los transectos.

6.- RESULTADOS OBTENIDOS




Especies de estudio. De izquierda a derecha: *Pseudalsophis dorsalis y Conolophus subcristatus*.

CULEBRAS

En total se capturaron, midieron y pesaron 15 culebras (ver Tabla 1). Ademas se encontro una culbera muerta, misma que ha sido depositada en la colección de vertebrados de la Estacion Cientifica Charles Darwin. Se colectaron muestras de tejido de 8 individuos vivos y de el especimen encontrado muerto (9 muestras total) acuerdo a protocolos aprobados por la Direccion General del Parque Nacional Galapagos.

IGUANAS

En total fueron capturadas y medidas 33 Iguanas, ver Tabla 2. A 31 de las mismas se les inserto un PIT TAG de acuerdo a protocolos aprobados por la Direccion General del Parque Nacional Galapagos. Dos de los individuos capturados representan recapturas.

DIRECCION DEL PARQUE NACIONAL GALAPAGOS

INFORME DE CAMPO





Personal de DGPNG muestreando culebras en Seymour Norte (Izquierda) e insertando un PIT en una Iguana terrestre (Derecha)

Tabla 1. Datos morfometricos de las culebras capturadas durante el periodo de monitoreo.

Fecha	Isla	*Hora de captura*	Codigo	Especie	Ventrales	Sub-caudales	Longitud total	Peso
11 Agosto 2017	Seymour Norte	8:15	0007922B95	*Pseudalsophis dorsalis*	230	110	73.5	40
11 Agosto 2017	Seymour Norte	9:11	0007922DD2	*Pseudalsophis dorsalis*	232	58	84.7	120
11 Agosto 2017	Seymour Norte	9:30	0007922B71	*Pseudalsophis dorsalis*	230	117	79.7	136
11 Agosto 2017	Seymour Norte	9:45	0007923CBF	*Pseudalsophis dorsalis*	195	104	67.6	43
11 Agosto 2017	Seymour Norte	11:36	0007922D03	*Pseudalsophis dorsalis*	234	114	68.7	122
11 Agosto 2017	Seymour Norte	12:49	0007922B36	*Pseudalsophis dorsalis*	226	96	73	55
11 Agosto 2017	Seymour Norte	14:58	0007922E	*Pseudalsophis dorsalis*	232	97	86.2	76
11 Agosto 2017	Seymour Norte	15:12	0007923BA6	*Pseudalsophis dorsalis*	232	81	90.7	101
11 Agosto 2017	Seymour Norte	15:53	0007923DD2	*Pseudalsophis dorsalis*	230	112	111.9	156
11 Agosto 2017	Seymour Norte	15:53	0007922DF0	*Pseudalsophis dorsalis*	230	47	75	59
11 Agosto 2017	Seymour Norte	16:18	SN10	*Pseudalsophis dorsalis*	230	122	56.9	102
11 Agosto 2017	Seymour Norte	16:30	0007922C78	*Pseudalsophis dorsalis*	234	94	93.3	100
11 Agosto 2017	Seymour Norte	16:45	0007922BBF	*Pseudalsophis dorsalis*	231	63	100	123
11 Agosto 2017	Seymour Norte	16:45	0007922ECB	*Pseudalsophis dorsalis*	230	106	104.1	144
11 Agosto 2017	Seymour Norte	17:02	0007923D72	*Pseudalsophis dorsalis*	229	114	77.3	50

DIRECCION DEL PARQUE NACIONAL GALAPAGOS

INFORME DE CAMPO

Tabla 2. Llista de Iguanas capturadas

Fecha	Hora de captura	Nueva/Recaptura	Locality	Brand	PIT	Sex	Largo cuerpo (cm)	Largo cola (cm)	Peso (kg)
14/08/17	10:57	Nueva	Seymour Norte	no	981098102929828	male	60.3	64.2	8.6
14/08/17	11:01	Recaptura	Seymour Norte	no	41838073	male	55.8	57	7.5
14/08/17	11:14	Nueva	Seymour Norte	no	981098102932893	female	49.5	53.3	4.7
14/08/17	11:25	Nueva	Seymour Norte	no	981098102934455	male	50.7	52	6.2
14/08/17	11:32	Nueva	Seymour Norte	no	981098102934244	female	49.5	59.8	4.3
14/08/17	11:40	Nueva	Seymour Norte	no	900026000577595	male	36.1	40.3	1.7
14/08/17	11:53	Nueva	Seymour Norte	no	900026000577600		30.5	36.1	1.1
14/08/17	12:07	Nueva	Seymour Norte	no	900026000577589		26.5	29.5	0.7
14/08/17	12:22	Nueva	Seymour Norte	no	900026000577579		23.5	28.4	0.5
14/08/17	7:25	Nueva	Seymour Norte	no	900026000577663	female	48	52	4.7
14/08/17	7:38	Nueva	Seymour Norte	no	900026000577665	female	45	53.8	5.6
14/08/17	7:52	Nueva	Seymour Norte	no	900026000577676	female	44	47	3.9
14/08/17	8:01	Nueva	Seymour Norte	no	900026000577664	female	38	44.8	2.5
14/08/17	8:08	Nueva	Seymour Norte	no	900026000577669	female	38.2	42.2	2.3
14/08/17	8:17	Nueva	Seymour Norte	no	900026000577629	female	37.2	42.4	2.1
14/08/17	8:34	Nueva	Seymour Norte	no	900026000577635	male	56.7	58	8.3
14/08/17	8:39	Nueva	Seymour Norte	no	900026000577650	male	57.5	54	6.8
14/08/17	8:46	Nueva	Seymour Norte	no	900026000577636	male	56.6	64.4	8.3
14/08/17	8:50	Nueva	Seymour Norte	no	900026000577651	male	48.3	43.5	6.5
14/08/17	8:56	Nueva	Seymour Norte	no	900026000577638	male	55.5	59	6.5

DIRECCION DEL PARQUE NACIONAL GALAPAGOS

INFORME DE CAMPO

Tabla 2. Continuacion

Fecha	Hora de captura	Nueva/Recaptura	Locality	Brand	PIT	Sex	Largo cuerpo (cm)	Largo cola (cm)	Peso (kg)
14/08/17	9:09	Nueva	Seymour Norte	no	900026000577634	male	59	53.3	7.9
14/08/17	9:19	Nueva	Seymour Norte	no	900026000577645	male	44.5	50.7	4
14/08/17	9:20	Nueva	Seymour Norte	no	900026000577786	male	52	52.2	5.3
14/08/17	9:38	Nueva	Seymour Norte	no	900026000577802	female	39.5	44	2.5
14/08/17	12:15	Nueva	Seymour Norte	no	900026000577580	male	55	55	6.5
14/08/17	13:18	Nueva	Seymour Norte	no	981098102931348	male	45	52.9	3.8
14/08/17	13:20	Nueva	Seymour Norte	no	900026000577781	female	51.5	51.5	4.5
14/08/17	13:33	Nueva	Seymour Norte	no	900026000577788	male	54	54.5	6
14/08/17	13:39	Nueva	Seymour Norte	no	900026000577800	male	53	54	7.2
14/08/17	14:08	Nueva	Seymour Norte	no	981098102933526	male	50	56.2	7.2
14/08/17	14:45	Nueva	Seymour Norte	no	981098102929899		21.8	27.5	0.4
14/08/17	14:45	Nueva	Seymour Norte	no	981098102935639		53	56.2	7.8
14/08/17	14:48	Recaptura	Seymour Norte	no	14562344	male	63.9	61.5	8.4
14/08/17	14:48	Nueva	Seymour Norte	no	981098102928884	female	49.5	51	4.8

DIRECCION DEL PARQUE NACIONAL GALAPAGOS

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Durante los transectos de DISTANCE se detectaron un total de 75 Iguanas en un transecto de 1.8 km, la distancia promedio de deteccion equivale a 17.5 m (rango: 0-45 m). El area cubierta por el transecto equivale a 13 hectareas (Figura 1). Estos registros dan como estimado una densidad promedio de 29.7 Iguanas por hectarea con minimo de 23.7 y maximo de 37.22. Este modelo preliminar sugiere que en el area de la plataforma habria 980 Iguanas, con extremos minimo y maximo de 782 y 1228 Iguanas, asumiendo una distribucion homogenea en el habitat. En este viaje no tuvimos tiempo de hacer un muestreo de DISTANCE en el area rocosa de la isla para calibrar esos estimados por lo que no es posible extrapolar al area total de la isla. En los proximos meses se realizara una estimacion general de Iguanas muestreando en todos los habitats presentes en Seymour Norte.



Figura 1. Seymour Norte, mostrando el area donde se efectuo el muestreo DISTANCE y los puntos en donde fueron registradas Iguanas Terrestres. El analisis preliminar de los datos indica una densidad de approximadamente 30 iguanas por hectarea en el area de La Plataforma.

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7.- CONCLUSIONES

- Se dio cumplimento al cronograma previsto de actividades, básicamente sobre el monitoreo del estado poblacional de las Iguanas Terrestres y colecta de muestras de tejido de Culebras.

8.- OBSERVACIONES

1. El trabajo de campo, fue financiado por Galapagos Conservancy, Galapagos Conservation Trust y Massey University.

9.- RECOMENDACIONES

1. Los datos obtenidos durante este monitoreo satisfacen las necesidades del programa de monitoreo anual de las especies reportadas.

Informe Elaborado por:

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Luis Ortiz-Catedral / Massey University

Aprobado por:

Danny Rueda y Christian Sevilla

MASSEY UNIVERSITY
COLLEGE OF SCIENCES

Auckland, Nueva Zelandia Mayo 13, 2018.

Danny Rueda Córdova

Christian Sevilla

Dirección de Ecosistemas
Proceso de Conservación y Restauración de Ecosistemas Insulares
DIRECCIÓN DEL PARQUE NACIONAL GALÁPAGOS

Estimados señores Rueda y Sevilla:

Reciban un cordial saludo. A través de la presente quiero darles a conocer información puntual acerca de la comunidad de reptiles en la Isla Seymour Norte y de *Opuntia echios* en la misma localidad, resultado de cuatro visitas a campo entre los meses de Agosto-Diciembre 2017 y Enero-Febrero 2018.

Tamaño poblacional de Iguana Terrestre (*Conolophus subcristatus*)

En base a un análisis de muestreo utilizando la metodología de "DISTANCE SAMPLING" para transectos, se estima que en la zona de mayor densidad de Iguanas terrestres, existen entre 1038 y 1275 individuos adultos, en un área de 21 hectáreas en la parte noreste de Seymour Norte. En el resto de la isla, las iguanas se encuentran a una menor densidad, y alcanzarían un tamaño poblacional entre 4242 y 5211 individuos adultos en un área total de 190 hectáreas. Esta metodología de muestreo subestima la densidad de juveniles o iguanas muy pequeñas, ya que estas tienden a esconderse entre la vegetación a medida que los observadores avanzan. Nuestras observaciones de Marca-Recaptura indican que las iguanas juveniles representan aproximadamente un tercio de la población en el área de alta densidad y un 15% en general en el resto de la isla. Ajustado las estimaciones de DISTANCE para reflejar la presencia de esta clase de edad, tendríamos una población total de iguanas de entre **5091** y **6253** individuos.



Arriba: Equipo en campo muestreando Iguanas en Seymour Norte.

Nivel de herbívoria en *Opuntia echios*

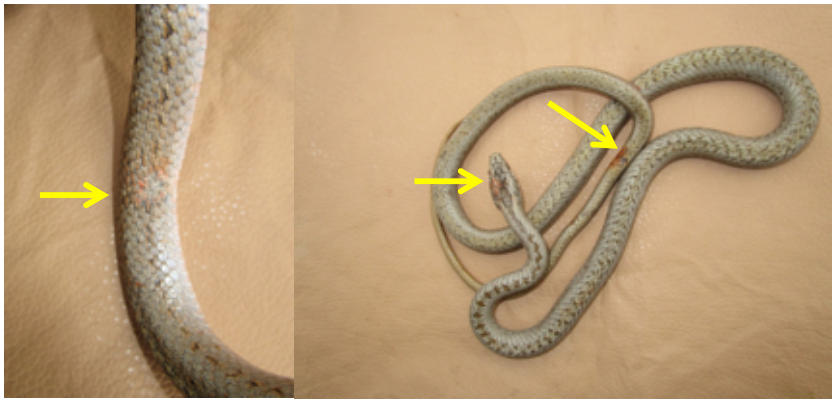
En Febrero del presente, realizamos una evaluación del nivel de herbívoria en *Opuntia echios* en el área de mayor densidad de Iguanas terrestres. En breve, asignamos una categoría de nivel de depredación en base al aspecto de los tallos en un transecto de 1 km. En este transecto encontramos un total de 380 plantas adultas, de las cuales el 70% fueron clasificadas como “depredación alta”, que indica que aunque las plantas están vivas, no tienen cladodios (tallos) fotosintéticos visibles (ver imagen). El resto de las plantas de *Opuntia* presento cladodios fotosintéticos. Cabe resaltar que no se registro ninguna planta juvenil de semilla durante nuestro muestreo.



Ejemplos de niveles de herbívoria en *Opuntia echios.* Izquierda, planta adulta con cladodios fotosintéticos presentes (depredación baja); Derecha, planta adulta sin cladodios fotosintéticos visibles (depredación alta).

Culebras terrestres

En Seymour Norte, habitan dos especies de culebra terrestre: La corredora pintada (*Pseudalsophis steindachneri*) y la corredora de Santa Cruz (*Pseudalsophis dorsalis*). A la fecha se ha encontrado un espécimen único de corredora pintada. La corredora de Santa Cruz, se encuentra en alta densidad en la parte noreste de la isla. A la fecha se han capturado, pesado, medido y fotografiado 51 individuos. De estos, el 40% muestra marcas a lo largo del cuerpo, atribuibles a mordeduras de roedor (ver imagen). Cabe destacar que la examinación detallada de los especímenes de corredora de Santa Cruz, siguieren que en realidad se trate de una especie nueva. En particular, la coloración (del gris pálido al rosa claro), el patrón de manchas dorsales y las series de escamas de las culebras de Seymour Norte, no concuerdan estrictamente con el material tipo de la especie. Actualmente se están realizando análisis moleculares en la Universidad de California, para determinar si en realidad esta es una especie nueva.



Corredora de Santa Cruz (*Pseudalsophis dorsalis*) mostrando daños atribuibles a roedores (flechas amarillas).

Quedo a su disposición. Por favor no duden en indicarme si necesitan información adicional.

Saludos cordiales

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