# Kent Academic Repository

**Turner, Helena Summers (2018) *Population Status and Conservation of the Critically Endangered Bermuda Skink.* Doctor of Philosophy (PhD) thesis, University of Kent,.**

# Population Status and Conservation of the Critically Endangered Bermuda Skink

Heléna Summers Turner

A thesis submitted for the degree of Doctor of Philosophy in
Biodiversity Management

Durrell Institute of Conservation and Ecology
School of Anthropology and Conservation
University of Kent

September 2018

Word Count: 52,608

# Population Status and Conservation of the Critically Endangered Bermuda Skink

Heléna Summers Turner

Supervised by:

**Professor Richard A. Griffiths**
Durrell Institute of Conservation and Ecology (DICE),
School of Anthropology and Conservation,
University of Kent, Canterbury, UK

**Dr. Gerardo Garcia**
Chester Zoo,
Cedar House, Upton by Chester, Chester, UK

**Dr. Mark E. Outerbridge**
Department of Environment and Natural Resources,
Government of Bermuda, Bermuda

## AUTHOR'S DECLARATION

I, Helena S. Turner declare that this thesis has been composed solely by myself with the incorporation of suggestions, feedback and editorial amendments made by Richard A. Griffiths, Mark E. Outerbridge and Gerardo Garcia, and that it has not been submitted, in whole or in part, in any previous application for a degree. Except where states otherwise by reference or acknowledgment, the work presented is entirely my own. The text does not exceed 100,000 words and meets the formatting guidelines of the University of Kent.

**Chapter 1:** HST wrote the chapter. Comments and feedback were provided by RAG.

**Chapter 2:** Analysis and writing: HST; experimental and statistical design: HST and RAG; field work: HST, RAG, MEO and GG; revisions: RAG, GG and MEO. All authors contributed critically to the drafts and gave final approval for publication. This chapter was published in *Oryx*.

**Chapter 3:** Analysis and writing: HST; experimental and statistical design: HST and RAG; field work: HST, RAG, and MEO; revisions: RAG, GG and MEO.

**Chapter 4:** Analysis and writing: HST; experimental and statistical design: HST and RAG; field work: HST, RAG, MEO and GG; revisions: RAG. Additional data provided by R. Wingate, A. Raine, and M.P. Hammond.

**Chapter 5:** Analysis and writing: HST; experimental and statistical design: HST and RAG; field work: HST, RAG, MEO and GG; revisions: RAG. Additional data provided by R. Wingate, A. Raine, M.P. Hammond, J. Davenport, A. Glasspool and M. Outerbridge.

**Chapter 6:** HST wrote the chapter. Comments and feedback were provided by RAG.

Additional comments were provided by the thesis examiners.

I hereby declare that there were no competing interests on behalf of all co-authors.

Any associated dataset and code will be made available through online repositories at the time of chapter publications.

Signed: Date: 20/09/20

## ACKNOWLEDGEMENTS

I would like to express my gratitude to all those who have supported and helped me during my PhD study.

My deepest gratitude goes first and foremost to Professor Richard Griffiths, my supervisor, for his patience, encouragement and guidance to help me grow as a researcher. I would also like to thank Dr. Gerardo Garcia as without his support my PhD would never have taken shape and thanks to you, I had enough data for a second PhD with all your extra suggestions.

A special thank you to Dr. Mark Outerbridge and Jeremy Maderios without whom my fieldwork would not have been possible. I am grateful for all the time you spent sharing your knowledge about Bermuda, and the valuable suggestions on my study. I am also deeply grateful to you and the Government of Bermuda's Department of Environment and Natural Resources for accommodation on Nonsuch Island for the duration of my fieldwork, for transport into the field and allowing me to continue collecting data (even after losing two boats, destroying a kayak and getting stranded on Horn Rock). I have no doubt Jeremy will continue to tell tour groups on Nonsuch Island about the 'crazy skink researcher!'

I am also thankful to Alison Copeland, Miguel Meijas and Nesta Wellman who were great hosts while in Bermuda and helped me out on numerous occasions. I would particularly like to express my sincere gratitude to Rebecca, Felicity, William, Rachel and Ubu Lawrence whom I cannot thank enough for their kindness and welcoming me into their home, and to Lyn Pantry for that time I was in desperate need of a hot shower and pizza.

And finally a massive thank you to everyone that assisted me in the field (Mark Outerbridge, Alison Copeland, Jeremy Maderios, Richard Griffiths, Gerardo Garcia, Miguel Meijas, Nesta Wellman, Jim Labisko, Stephanie Clayton-Green, Dylan Muldoon, Nathan Wright, Luiza Passos, Dario D'Afflitto, Jack Carney, Mandy Shailer, Peter Drew, Monika Alonso, Russ Frith, Kirk Trott, Robin Marirea, J.P. Rouja, Phillipe Rouja, Lynne Thorne) that made the boating adventures, long hours of extreme heat and humidity, rotten fish and prickly pear spikes bearable.

I wish to thank Professor Jim Groombridge and all my DICE colleagues who took the time to help me and provide advice: Izabela Barata, Andrew Buxton, Debbie Fogell, Hazel Jackson, Simon Tollington, Jim Labisko and Jamie Sells. I am also grateful to Dr. Rachel McCrea, Dr. Diana Cole, Dr. Eleni Matechou who assisted in data analysis and modelling construction during various statistics workshops. I would additionally like to thank John Davenport and Zoe G. Davies for their valuable comments and recommendations.

Lastly my thanks would go to my wonderful family for their whole-hearted support for the past four years. I couldn't have done it without you all!

This research was supported by several funders and collaborators that I am extremely grateful to June Summers Shaw, Chester Zoo, The States of Jersey, the British Herpetological Society and the Bermuda Zoological Society that supported this study. Thanks also to the Government of Bermuda's Department of Environment and Natural Resources, Professor Richard Preziosi at Manchester Metropolitan University (previously at the University of Manchester) and Dolphin Quest for providing additional field supplies and equipment.

*(Photo credit: Turner, H. 2016)*

# ABSTRACT

Global biodiversity is currently facing the sixth mass extinction with species disappearing at a rate 100 times background levels - mostly driven by anthropogenic pressures. Island species are often evolutionarily distinctive and are highly vulnerable to novel disturbances due to having small geographic ranges. The isolation of Bermuda has led to the evolution of a unique ecosystem with many endemic species. However, native species across Bermuda have become especially vulnerable ever since human colonisation, due to habitat loss and destruction, and predation and competition by several introduced species. The Bermuda skink (*Plestiodon longirostris*) was once widespread and commonly found along stone walls and cedar groves across Bermuda. By 1965, the population had become extremely fragmented and was declining across Bermuda and its offshore islands.

This thesis investigates the threats to the Bermuda skink, listed as Critically Endangered by the International Union for the Conservation of Nature. Surveys were conducted between 2015 and 2017 across Bermuda to (1) estimate population parameters such as abundance, capture and survival probabilities; (2) estimate occupancy, colonisation, extinction and detection rates; (3) determine if skink sub-populations are morphologically different; and (4) compare the body condition between sub-populations that may provide warning signs of issues in the environment such as changes in the level of competition, predation or available habitat. Additionally, recommendations are provided for the conservation management and recovery of this species.

From capture-mark-recapture surveys and subsequent robust design modelling, it was found that the two largest populations fluctuated in size at both sites over the three-year survey period, and are imminently threatened by increasing anthropogenic activities, invasive species, and habitat loss in Bermuda. Using dynamic occupancy modelling across Bermuda, skinks were detected at 13 locations. The probability of detection was higher on island sites and with the presence of seabirds, prickly pears and coastal habitat. However, skinks were unlikely to be detected at sites with cat and rat predators. We demonstrate that morphological diversification has occurred, possibly in response to isolation and changes to habitat and predator levels over time, especially on these small offshore islands. Finally, our study showed that trends in body condition differed between sub-populations with the two largest sub-populations – on Castle and Southampton Islands – having higher body condition compared to other populations. Overall, body condition has declined significantly over the past 15 – 20 years.

After more than 50 years of study of the skinks, there is sufficient evidence to identify the reasons for their population decline. Although future studies may be needed to monitor populations, long-term the ongoing threats these populations face should be mitigated to help prevent extinction.

# LIST OF FIGURES

# LIST OF TABLES

# TABLE OF CONTENTS

# CHAPTER 1

# GENERAL INTRODUCTION

Bermuda archipelago.
(*Photo credit: Skinner, R. 2014*)

## 1.1 GLOBAL BIODIVERSITY CONSERVATION

Global biodiversity is currently facing the sixth mass extinction, with species disappearing at least 100 times the expected natural rate (Ceballos et al., 2017). According to the International Union for Conservation of Nature (IUCN), which is recognised as the most comprehensive inventory evaluating the threat status of various plant and animal species, the home ranges of more than 27,600 species of land vertebrates have shrunk by a third, between 1900 and 2015, with more than an eighth of all birds, a quarter of all mammals, a fifth of all reptiles, a third of all amphibians now known to be in a perilous position, ultimately caused by human overpopulation and continued population growth (Ceballos et al., 2017; IUCN, 2018a). Studies worldwide increasingly demonstrate human-induced global environmental changes influencing wildlife, particularly through habitat loss linked to agricultural changes, urbanisation and industrial development (e.g. Brashares et al., 2001). In total, about 80% of all the forests that covered the Earth 8,000 years ago have been cleared or degraded by man, and by 2030 it is estimated only 10% will remain (Nielsen, 2006).

Global conservation efforts have created more than 44,000 wildlife reserves as well as preventing the extinction of several species. Notable successes in individual conservation projects include bringing Golden lion tamarins (*Leontopithecus rosalia*), Black-footed ferrets (*Mustela nigripes*), American bison (*Bison bison*), and Mauritius kestrels (*Falco punctatus)* back from the brink of extinction, through concerted captive breeding programmes and reintroducing individuals back to the wild (Jones et al., 1994; Dobson and Lyles, 2000; Hedrick, 2009; Kierulff et al., 2012). Local extinctions result from threats to their habitats, such as over-exploitation (i.e. poaching, pet trade), habitat loss, disease, pressures from competition from introduced (or alien) animals and plants. Additionally, climate change is becoming an increasingly significant problem (Atkinson, 1996; Burns et al., 2003; Rodrigues, 2006; Smith et al., 2006; Zhang et al., 2008; Bouché et al., 2012). As the majority are human-driven, human actions alone can therefore prevent many species from becoming extinct.

Islands are biodiversity hotspots that contribute to the maintenance of many ecosystem functions such as providing defence against natural disasters, supporting nutrient cycling, and soil and sand formation (Whittaker and Fernandez-Palacios, 2007). The importance of islands in revealing evolutionary processes has been recognized since Darwin's work on the Galapagos (Darwin, 1859; 1909) and Wallace's work in the Malay Archipelago (Wallace, 1876). Because isolation is a driver of speciation, island species display an astonishing variety of traits and species are often found nowhere else in the world. Island biogeography has provided many examples of the evolutionary mechanisms involved in generating biodiversity, including geological processes and colonisation and isolation (Emerson, 2008; Gillespie et al., 2008; Parent et al., 2008). Essentially, island biogeography theory predicts that larger islands will have a greater number of species than smaller islands, and the probability of colonisation of large islands will be higher than that of small islands. Thus, the larger the island and closer it is to the mainland, the more potential species will arrive. Species will also more readily go extinct on small islands than large, due to factors such as smaller population sizes and less available habitat (MacArthur and Wilson, 2001). This simple model is quite powerful and remains fundamentally important in understanding the effects of habitat fragmentation and predicting species richness to assist with conservation efforts, which are often neglected within oceanic islands (Kier et al., 2009).

The potential consequences of global climate change for animal populations have emerged as ranking amongst the greatest perceived threats to biodiversity (e.g. Hughes, 2000; McCarty, 2001; Walther et al., 2002). There have been significant periods of global warming and cooling in the Earth's past, with higher than average global temperatures compared to current temperatures (Zachos et al., 2001). Many species have adapted to and survived through these events (Huber, 2009), but it is the rapid rate at which temperature is increasing today which is unprecedented (Houghton, 1997; Root et al., 2003). With this rate of change in environmental conditions it will be beyond the ability of many species to adapt to or evolve (Markham, 1996; Kingsolver, 2009) and this will have implications for biodiversity.

While both mainland and islands suffer from several major threats to biodiversity the current threat is significantly higher on islands (Frankham, 1998; Bryant, 2002; Clout and Veitch, 2002). Sea-level rise remains one of the least studied of all climate-change issues (Courchamp et al., 2014). Many low-lying islands will simply be entirely submerged, threatening island biodiversity and any benefits from recent conservation actions (Glasspool, 2008). A recent analysis focusing on 4,500 islands in ten biodiversity hotspots suggested that 6–19% of these islands could be entirely submerged with a 1–6 m sea-level rise, threatening over 300 endemic species with extinction (Bellard et al., 2013). Islands also have the highest proportion of recorded species extinctions, 80% of known species extinctions have occurred on islands and currently 45% of IUCN Red List endangered species occur on islands (IUCN, 2018b). Before human colonisation, New Zealand had 94 species of native land birds; 35 were exterminated prehistorically and 8 more have become extinct in the historical period. Hence 46% of the original bird fauna is now extinct, including all 11 species of large flightless birds called Moas (Bryant, 2002). Not only were the flightless birds probably easy prey for humans, but the dense human population also cleared large areas of forest and introduced pigs (*Sus scrofa domestica*), domestic dogs (*Canis familiaris*), and chickens (*Gallus domesticus*), which may have additionally introduced avian diseases to the native birds.

After the Polynesians arrived in the Hawaiian Islands in the 13th century, a wave of avian extinctions occurred, where 51% of native land birds became extinct. This may have been largely due to the Polynesians accidentally introducing rats (*Rattus exulans*), which may have fed on the eggs and chicks of ground-nesting birds (Bryant, 2002). These novel disturbances decimate native island wildlife and change entire island ecosystems. It is estimated that between 5% and 20% of all invasive species are regarded as a major threat to native ecosystems and to global biodiversity (IUCN, 2018c).

## 1.2 GLOBAL REPTILE DECLINES

Our state of knowledge for the conservation status of reptiles is much less complete than for other taxa. Of the 10,973 species of known reptiles, only 1,437 have sufficient data to have been evaluated for conservation status. That leaves 87% of reptiles for which we lack sufficient data to know whether their populations are fluctuating or remain stable. Furthermore, the means of determining a species' conservation status is a rigorous and time-consuming process, and therefore many threatened species may be underestimated due to data deficiency. While effective conservation measures are increasing populations of some species, gaps in knowledge need to be addressed to ensure that reptiles continue to thrive around the world as the current levels of conservation are insufficient (Hoffmann et al., 2011).

Reptiles and amphibians are recognised as integral parts of natural ecosystems and as indicators of environmental quality (Gibbons and Stangel, 1999). Although the amphibian decline problem is a serious threat, reptiles appear to be in even greater danger of extinction worldwide (Gibbons et al., 2000). Despite conservation measures to save them, hundreds of reptile species are expected to become extinct over the next century. Six significant threats to reptile populations established by (PARC) Partners in Amphibian and Reptile Conservation (Gibbons and Stangel, 1999) are: habitat loss and degradation , introduced invasive species, environmental pollution, disease, unsustainable use and global climate change. An additional category comprises unexplained declines. For example, thousands of reptiles are killed by traffic on roads every day (Aresco, 2005; Van der Ree, 2015) and many populations have become locally extinct through the flooding of valleys as a result of heavy rainfall or dam construction projects throughout the world (Cooper, 1984).

Reptiles are also hunted for food, their eggs, or the pet trade (Spotila et al., 1996; Gibbons and Stangel, 1999). In particular, the harvesting of sea turtles and their eggs caused a significant decline in many nesting populations around the world (Pritchard, 1980; Tomillo et al., 2008; Tapilatu et al., 2013). During this time, nearly all loggerhead turtle (*Caretta caretta*) eggs laid along the Brazilian coast were removed, and most nesting females were taken for meat (Marcovaldi et al., 2005; Marcovaldi and Chaloupka, 2007). Despite international protection, sea turtles are still at risk of global extinction (Dutton et al., 2005) as poaching still continues (Koch et al., 2007), and in more recent years, they have become exposed to other hazards such as marine debris, pollution (Bugoni et al., 2001) and fishery-induced mortality (Soto et al., 2003, Kotas et al., 2004; Koch et al., 2007).

Even some species that were common and widespread a few decades ago are becoming scarce. For example, forest skinks (*Emoia nativitatis*) have gone from being abundant to absent across the rainforests of the 135 km$^2$ Christmas Island. In 1998, more than 80 individual forest skinks were recorded basking and foraging around a single fallen tree. Then began a rapid and apparently inexorable decline. By 2003, they were confined to scattered pockets in remote parts of the island. By 2008, a targeted survey found them at only one remaining site. Now, recent repeated searches and trapping have failed, and the species appears to have disappeared completely from its natural habitat (Smith et al., 2012). Five of the six reptile species on Christmas Island have now declined to near extinction and it is likely that predation by introduced species is the major factor (Smith et al., 2012).

Regrettably, many reptile populations that are thought to be declining have not been monitored over long periods of time, making short-term changes in population size difficult to evaluate critically. Among reptiles, declines in sea turtles are perhaps the best documented because populations of sea turtles engage in site-specific annual breeding (Bolten and Witherington, 2003). In contrast the majority of reptile species do not congregate to breed or hibernate and do not occur at particularly high densities within their habitats. Population declines can be difficult to detect; hence, long-term studies of natural populations and communities are fundamental to understanding natural population trends and fluctuations (Tinkle, 1979). Natural population fluctuations and local extinctions are common in reptiles especially on islands (Case et al., 1992) and generally are no cause for alarm, but not all declines are natural (Gibbons et al., 2000). As of 2017, the IUCN listed 31 species of reptile that have become extinct in recent years. Twenty-nine of these lived on islands. (Maas, 2017; IUCN, 2018a). Island species are especially vulnerable because their environment is affected by human impact and by the introduction of predatory animals. For example, all eight endemic reptiles went extinct due to the impact of invasive species on the island of Rodrigues in the Indian Ocean (Cheke and Hume, 2008).

### *1.2.1 Global skink declines*

Skinks are among the most threatened reptile species and are at a high risk of extinction (Purvis et al., 2000). In part, this is because they have small geographical ranges, and are often endemic and only found on small oceanic islands (e.g. Otago and Grand skinks *Oligosoma otagense* and *O. grande* respectively, in New Zealand, Bojer's skink *Gongylomorphus bojerii*, Telfair's skink *Leiolopisma telfairii*, Macchabé skink *Gongylomorphus fontenayi* and Orange tailed skinks *Gongylomorphus* spp. in Mauritius, blue tailed skinks *Cryptoblepharus egeriae* of Christmas Island and prehensile-tailed skinks *Corucia zebrata* in the Solomon Islands). As skinks are generally cryptic, rare, secretive, or are burrowing species they tend to be difficult to study and are therefore poorly understood (Beheregaray and Caccone, 2007).

In 2002, 24 new species of skinks, all from islands in the Caribbean, were discovered and scientifically named through examination of museum specimens (Hedges and Conn, 2012).

However, all of the newly identified endemic Caribbean skinks are near extinction (or are already extinct) due to introduced predators including mongooses (*Herpestes javanicus*) and cats (*Felis catus*), as well as large-scale habitat destruction due to development and agriculture (Hedges and Conn, 2012). This loss is alarming because reptiles play a number of important roles as predators, prey and seed or fungal spore dispersal in their ecosystems and are consequently valuable indicators of environmental health (Heatwole, 1976; Hartup, 1996; Burger and Gibbons, 1998; Aguire and Lutz, 2004; Beaupre and Douglas, 2009; Cooper and Viernes, 2011; Valencia-Aguilar et al., 2013). Reptiles are often associated with extreme habitats and extreme environmental conditions, so it is easy to assume that they will be resilient in a changing world. However, many species are highly specialised in terms of habitat use and the climatic conditions they require. This makes them particularly sensitive to environmental changes (Hedges and Conn, 2012).

## 1.3 BERMUDA

Bermuda lies at 32.20°N and -64.45°W and is a British overseas territory situated in the Atlantic Ocean 1,052 km east of Cape Hatteras, North Carolina, United States which is the closest point of land (Procter and Fleming, 1999; Fig. 1.1).

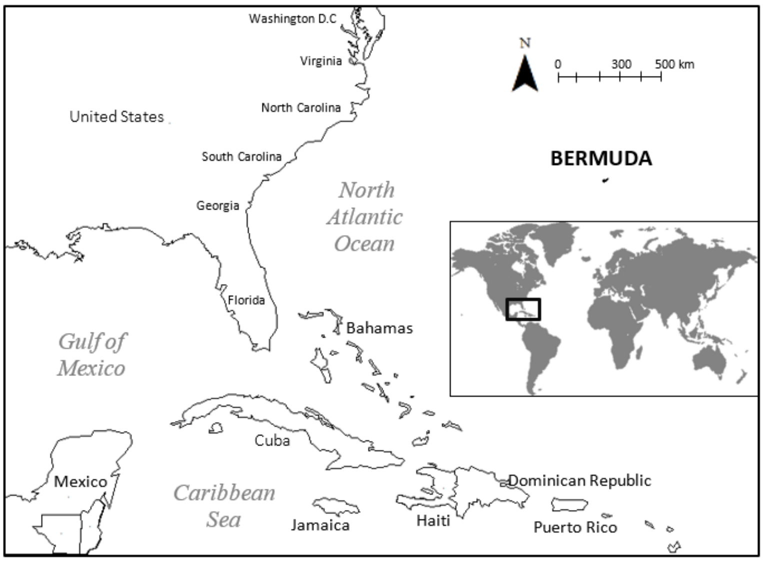


**FIGURE 1.1**. Location of Bermuda in the North Atlantic Ocean (Image created in ArcMap 10.5).

The Bermuda ecoregion is a crescent-shaped chain with approximately 193 islands and islets as a result of extensive volcanic activity along the mid-Atlantic ridge from 110 million years ago (mya) establishing a series of volcanic peaks (Fig. 1.2). They ceased activity around 33 mya (Ellison, 1996; Copeland, 2011).

**FIGURE 1.2**. Evidence of Bermuda's volcanic past: an aerial view of two calderas within the Great Sound and Castle Harbour, Bermuda (*Photo Credit: NOAA*).

Volcanic peaks gradually eroded below sea level by wave action, in the shallows. Limestone was deposited from the calcareous algae, resulting in a thick (15 - 100 m) limestone cap covering Bermuda, in which the limestone surface is overlain by a very shallow layer of fertile soil. As a result, the ecoregion has no rivers or surface streams and the shoreline is composed of bays and inlets, with coral sand beaches lining the shores, primarily in the south (Sterrer, 1998). As the Bermuda seamount is situated near the middle of the oceanic portion of the North American tectonic plate, Bermuda additionally experiences significant seismic activity. In the last 350 years, a total of 56 earthquakes have affected Bermuda (Brewer, 2014). Other evidence that Bermuda has been subjected to episodic instability comes from rock fractures which can range from mere cracks, to 0.75 m wide fissures across the islands (Rowe, 1998). Certainly, many of these fractures have locally been enhanced by coastal erosion, but to what extent is hard to assess. Either way, erosion of the sand dunes and limestone rock makes the coastal areas incredibly vulnerable.

Bermuda has a total land area of 55 km², stretching 55 km in length and 1.61 km across. It is moderately low lying, with the highest elevation reaching only 78 m (Copeland, 2011). Sixty-four of the Bermuda islands have a history of human habitation. Of these, eight of the main larger islands are currently connected by bridges or causeways (Copeland, 2011).

Bermuda holds the distinction of having passed the first conservation legislation in the New World (Sterrer, 1995), protecting the Bermuda petrel also known as the Cahow (*Pterodroma cahow*) and other birds as early as 1616 and limiting the uses of native cedar as early as 1622 (Rueger and von Wallmenich, 1996). A comprehensive and well-managed protected areas system currently exists, comprising 58 nature reserves that cover some 48 hectares, as well as 75 amenity parks and at least 8,299 recorded species, 4,597 of which are marine and 3,702 are terrestrial (Copeland, 2011). Of these 110 marine species (2.4%) and 137 terrestrial species (3.7%) are considered endemic (Sterrer, 1998; Copeland, 2011). Endemic species are especially vulnerable as Bermuda has already witnessed the extinction of at least 60 known species of endemic flora and fauna since the time of human settlement. These include: two Bermuda land snails (*Poecilozonites nelsoni* and *P. reinianus)* due to the introduction of carnivorous snails (*Euglandina rosea*) and flatworms (*Bipalium vagum*), the Bermuda cicada (*Tibicen bermudiana*) and the geometrical moth (*Semiothisa ochrifascia)* due to the near-eradication of the Bermuda cedar (Sterrer, 1998; Olsen et al., 2006) and the Bermuda tortoise (*Hesperotestudo bermudae*), however the cause of the species extinction is still undetermined as only two bone fragments have ever been found (Olson and Meylan, 2009). Among birds, at least ten presumed endemics are known from Pleistocene fossils. These include a crane (*Grus latipes*), a duck (*Anas paschyscelus*), a night heron (*Nyctanassa carcinocatactes*), three species of rail (*Rallus ibycus*, *Porzana piercei, Rallus recesss*), a woodpecker (*Colaptes oceanicus*), a hawk (*Bermuteo avivorus*), a heavy-billed passerine (*Pipilo naufragus*), and a small owl (*Aegolius gradyi*; Sterrer, 1998; Olson and Wingate, 2000; Olson, 2012). Most of these birds are thought to have gone extinct due to sea level changes (up to 21 meters above present levels; Olson and Wingate, 2000) before human settlement, although the passerine, heron and owl were present at the time of the first settlers (Sterrer, 1998). An estimated 16 insect species have disappeared in the last century, including the flightless grasshopper (*Paroxya bermudensis*), seven true fly (Diptera) species, and the only endemic nemertean (*Pantinonemertes agricola*) a type of ribbon worm (Sterrer, 1998). Several other endemic species remain on the brink of extinction including two further Bermuda land snail species (*Poecilozonites circumfirmatus* and *Poecilozonites bermudensis*), the Bermuda ant

(*Odontomachus insularis*), Bermuda buckeye butterfly (*Junonia coenia*), Bermuda killifish (*Fundulus bermudae*), Bermuda cave shrimp (*Mictocaris halope*) and the Bermuda petrel.

Bermuda's position as an isolated oceanic island has contributed to a flora and fauna quite distinctive from any continental area or from Neotropical islands to the south (due to prehistoric sea level fluctuations as well as a lack of anthropogenic influence prior to the 1600). As a result, Bermuda contains no native mammal or amphibian residents, although four species of visiting bats have been recorded. This has led to the evolution of some unique endemic species, one of those being the Bermuda skink *Plestiodon longirostris* previously known as *Eumeces longirostris* (Smith, 2005).

## 1.4 THE ORIGIN OF SKINKS IN BERMUDA

There are approximately 5,634 recognised species of lizards worldwide (Pincheira-Donoso et al., 2013). Of these, skinks number more than 1,230 described species and are currently the largest group of lizards (Utez et al., 2018). They are found on every continent, with the exception of Antarctica, and are most abundant in the tropics (Cash, 2015). Skinks are classified into a group of lizards belonging to the family Scincidae and the infraorder Scincomorpha (Cash, 2015), which includes spiny-tailed lizards (Agamidae), night lizards (Xantusiidae), wall lizards (Podarcis), spectacled lizards (Gymnophthalmidae) and whiptails (Cnemidophorus). Almost all skinks have smooth, shiny scales, long cylindrical bodies with no pronounced neck (Cash, 2015). They have moderately short limbs, but several genera have no limbs at all (e.g. *Typhlosaurus*), so their locomotion resembles that of snakes more than that of lizards with well-developed limbs (Cash, 2015).

The Bermuda skink was first described in the 1861 by the American herpetologist Edward Drinker Cope. More recently, the discovery and analysis of fossil bones indicates that skinks were living on Bermuda more than 400,000 years ago. Further paleontological and geological evidence suggest they may have been present for 1 – 2 million years from as early as the Pleistocene epoch (Olsen et al., 2006). Bayesian phylogenetic analyses also strongly suggests that Bermuda skinks are descended from a *Plestiodon* species that once inhabited the Eastern USA (Brandley et al., 2010), and it is believed to be the only representative of one of the oldest extant *Plestiodon* lineages diverging more than 12 million years ago (Brandley et al., 2012).

This implies that, within a short geological timeframe, mainland North American ancestors of Bermuda skinks colonised the recently emergent Bermuda and the entire lineage subsequently vanished from the United States (Brandley et al., 2010).

Skinks or their eggs may have arrived in Bermuda via oceanic currents on vegetation rafts from North America, the closest landmass to Bermuda (Raine, 1998). To date, evidence for successful rafting has been found in more than 1,200 species, including plants, invertebrates, insects, reptiles, amphibians and mammals (Morton, 2013). Specifically, during the Pleistocene era it has been documented that many skink species had dispersed by rafting on fallen branches, trees or mats of vegetation washed into the ocean during floods or hurricanes. For example, *Chioninia* species are thought to have travelled from west Africa to Cape Verde and between its Barlavento and Sotavento islands (Austin and Adler, 1995) and *Leiolopisma* species from New Caledonia or Australia to New Zealand (Kuschel, 1975). This shows that this type of migration is entirely possible. The skinks would have been well adapted to deal with the harsh salty and sunny conditions of weeks at sea with little to no food, surviving their crossings by entering a hibernation-like state to conserve energy (Morton, 2013). Records show that no mammal or amphibian species travelled further than 450 km from the mainland (Darwin, 1859) and Bermuda is over 965 km from North America. Distance therefore explains why there is an absence of native amphibians and terrestrial mammals on Bermuda, as it does not lie close enough to the continent for them to survive the challenging journey across the sea.

Because of its isolation, the Bermuda skink has retained various primitive morphological characteristics such as a long tapering head and body. It is also more snakelike than other *Plestiodon* species of eastern North America (Heilprin, 1889). Hence the name ‘longirostris’, meaning long snout in Latin. The skinks also have powerful jaws with many extremely small pleurodont teeth, which is where the genus name is derived from the Greek words *pleistos* meaning “many” and *odontos* meaning “teeth”.

## 1.5 BERMUDA SKINK LIFE HISTORY

The Bermuda rock lizard or skink is unique in being the only non-avian, endemic terrestrial vertebrate in Bermuda (Edgar et al., 2010). The Bermuda skink, a diurnal, medium sized scincid lizard [adult snout to vent length (SVL): 72 - 94 mm and 12 - 30 g body mass] was first described by Jones and Hurdis in Bermuda, defining the status of the skinks in the 1840s (Jones et al., 1859; Barnes and Edy, 1987). Despite once being common across Bermuda, relatively little is known of its ecology, behaviour, and general biology because of its secretive habits and semi-fossorial lifestyle.

Adult skinks are dark grey and their lower jaw edges and cheeks are mostly a salmon pink to orange colour. The dorsum of the sub-adults varies between shiny dark brown to grey body scales with two sets of dorsolateral black and white stripes extending from the head to the base of the tail. Juveniles have a very similar body colour to the sub-adults but with distinctive bright blue tails, which they can shed to distract predator attention away from the body (Heilprin, 1889; Davenport et al, 1997).

The skinks have short, stout legs with clawed feet and a long tapering tail. They are well adapted for burrowing, so will feed around the grasses, rock walls and narrow rock crevices. They mostly prey on small invertebrates such as ants (*Pheidole megacephala*), beetles (Coleoptera), cockroach nymphs (*Periplaneta americana*) and woodlice (*Armadillidium vulgare*; Wingate, 1965). However, they are also natural scavengers with a keen sense of smell that attracts them to carrion, unlike most reptiles (Garber, 1988). It is thought they have a mutualistic relationship with the white-tailed tropicbirds (*Phaethon lepturus catesbyi*) and Bermuda petrels (*Pterodroma cahow*): the skinks benefit from eating any broken eggs, dead chicks and dropped fish in the nests, while the birds benefit from skinks cleaning the nest ready for the next breeding season, thereby reducing parasites or diseases (Garber, 1988; Wingate, 1965). As they feed opportunistically, skinks have even been observed eating baygrape fruits (*Coccolbis uvifera*), prickly pear fruits (*Opuntia dillenii*) and appear to be very fond of cheese (Wingate, 1965; Davenport et al., 1997).

The skinks typically live within coastal habitats that include sand and dune, rocky coastal outcrops and upland coastal scrub habitat (*see Supplementary Material 4 for vegetation found within each habitat type*). Within these areas they are limited to patches with leaf litter, matted crab grasses (*Digitaria sanguinalis*), sea oxeye (*Borrichia frutescens*), seaside goldenrod (*Solidago sempervirens*), buttonwood (*Conocarpus erectus*) and bay grapes

(*Coccolbis uvifera*), that provide ideal ground cover with rich insect abundance and concealment from predatory birds (Wingate, 1965).

Bermuda skinks are active all year round with peak activity in the summer months, retreating into deep cliff crevices, rock walls or the ruins of old forts when the temperature drops below 21°C (Wingate, 1965). Equally, the skinks will avoid the heat of the day and exhibit crepuscular activity patterns. Breeding season begins in spring with males developing bright orange cheeks and lower jaws (Fig. 1.3) and exhibit territorial and aggressive behaviour (Pyron and Camp, 2007; Vitt and Cooper, 1986). There is strong evidence in other lizards that this conspicuous coloration developed during the breeding season is a sexual characteristic used in male-male competition (Cooper and Vitt, 1993) and individuals with more intense colouration or larger areas of colouration were more likely to have mating success (Salvador et al., 2013; Zucker, 1994; Olsson, 1994). Between May - June, females will lay three to seven eggs in deep rock crevices, hatching around 40 days later. She will guard the eggs but does not take care of young and will aggressively chase them away, thus dispersing them (Edgar et al., 2010). Skinks will reach sexual maturity between two and four years old (Edgar et al., 2010).

**FIGURE 1.3**. Differences in cheek colourations between adult Bermuda skinks. (*Photo Credit: Turner. H, 2015*).

As Bermuda skinks have never been studied long-term their life span still remains unclear. Although it was thought that some individuals from a toe-clipping study were still being observed fifteen years later (Glasspool and Outerbridge, 2004), it is more likely this was mistaken for natural toe loss. It is thought the Bermuda skinks have a limited home range of around 10 $m^2$ (Davenport et al., 1997). Similar to other *Plestiodon* species the Bermuda skinks home range probably varies in size and shape in accordance with the age and gender of the individual skink (Fitch, 1956).

## 1.6 CONSERVATION BACKGROUND

The skinks were described as 'very common' throughout Bermuda in the 1800s and were often found around old stone walls and rocks within the cedar groves (Jones et al., 1859; Cope, 1861). Verrill (1902) reported that skinks were 'by no means common', rarely seen on the mainland and were prevalent only on Castle Island around the forts and cliffs. The difference between these reports suggests that there was a notable decline of Bermuda skinks within four decades. However, it is possible that the skinks were never particularly conspicuous. Observations by Bermuda's Department of Agriculture began in 1956 and continued sporadically until 1963 when populations remained on the mainland particularly on the North Shore in Devonshire and the Castle Harbour Islands, but their distribution was patchy and local residents had noticed a dramatic decline in numbers (Wingate, 1965; Garber, 1988).

A population survey was undertaken on Nonsuch Island in 1970 and it was found that there was a large sub-adult and juvenile population. Again, in the 1990s the few remaining areas that skinks remained were largely dominated by old adults, signifying a high mortality of hatchlings and juveniles, possibly due to an increase in predation (Griffith and Wingate, 1994). As a result of the skink's current limited and patchy distribution, the concern for the skink's survival began and further action was taken (Griffith et al., 1991). The Bermuda skink was listed as Critically Endangered by the Species Survival Commission of the IUCN in 1996 (Conyers and Wingate, 1996; Baillie and Groombridge, 1996; IUCN, 2015). They receive the highest protection (Level 1) under Bermuda law since 2003 (Protected Species Act, 2003; 2011). Any offenders would be liable to a fine of $5,000, and up to $10,000 for continued offences, for wilful damage or destruction, removal of habitat, importing, exporting, selling, purchasing or having a protected species in possession.

Following this, two skinks were captured from Spittal pond nature reserve (located on the coast of Smith's Parish), the largest wildlife sanctuary in Bermuda and thought to hold the majority of the mainland population of skinks. The individuals were displayed at the Bermuda Aquarium, Museum and Zoo (BAMZ) in the ‘Local Tails’ exhibit providing an important educational attraction for the public as well as an opportunity to learn more about their natural history. However, the exhibit was removed in 2016, to make space for a new quarantine facility. By the end of 1998, a three-month campaign in partnership with John Barritt and Sons Ltd called ‘Have a drink, save a skink’ was launched in which all ginger beer cans displayed a picture of the Bermuda skinks with a message about the threat of leaving cans as rubbish (Fig. 1.4). Visitors bringing the can with them received half price admission to BAMZ. However, a small amount of negative feedback terminated this initiative, because a fear of lizards meant some people refused to buy the cans as they had a picture of a lizard on them (B. Barritt, pers. comm. 2015).

**FIGURE 1.4**. Bermuda skink campaign in 1999. A collaboration between Barritts Ginger Beer company and the Bermuda Aquarium, Museum and Zoo.

In the summer of 1998, a questionnaire designed by the Bermuda Zoological Society (BZS), was mailed to 33,000 households with support from the Bermuda Electric Light Company (BELCO), seeking island-wide information regarding the distribution of the skinks, including a picture of the lizard and asking where and when they had seen any. Over 150 people responded to the survey and showed areas where skinks still appeared to be present particularly along Bermuda’s south shore (Glasspool, 2000). These included Horseshoe Bay, Warwick Long Bay, and Astwood Park, but the results showed many isolated populations across the mainland (*See Supplementary Material 1 for map of skink sightings*). Although the low response rate may not have been a true reflection of the skink distribution, the sightings provided a starting point for future skink surveys, particularly on the mainland (*See Supplementary Material 2 for summarised data on skink surveys*).

Various population assessments have been undertaken over the past two decades (*See Supplementary Material 1 and 2*) with sub-populations (i.e. a subset of the population) of skinks being reported from at least 24 sites (i.e. a site being a local area where skinks can be found, often an island or nature reserve) across Bermuda (Edgar et al., 2010). The greatest concentration is found within Castle Harbour, (several islands that make up the protected nature reserve), where the largest sub-population is currently found there, on Southampton Island (estimated at 582 skinks; Glasspool and Outerbridge, 2004).

Throughout this thesis a ‘population’ refers to the whole Bermuda skink population, whereas a sub-population refers to a subset of a larger population. Multiple sub-populations across Bermuda also act as metapopulations which means various sub-populations interact at some level via dispersal. The concept of a metapopulation was introduced by Richard Levins in 1969. The current most popular approach ‘The Levins model’ or ‘The classical metapopulation model’ is based on a population with occupied and unoccupied patches and an adequate rate of migration (immigration into vacant patches and emigration from occupied patches) among patches (Levins, 1969; Harrison and Taylor, 1997). An important assumption is that all sub-populations have a significant risk of extinction but are sufficiently close to neighbouring sub-populations for dispersal and therefore recolonisation to occur (Stith et al., 1996).

Other metapopulation models include: (1) ‘The patchy metapopulation model’, where all sub-populations are sufficiently close to function as a single population. That means that enough individuals are migrating between sub-populations that the sub-populations are not at risk of going extinct (Stith et al., 1996). In this model, the sub-populations are not independent, and their demographics are closely linked and when a sub-population goes extinct it is not noticed because it is part of a large continuous metapopulation. However, if a single large sub-population acting as a ‘source’ goes extinct then the wider metapopulation may crash. (2) ‘The non-equilibrium metapopulation model’, where each sub-population acts as a separate metapopulation (Stith et al., 1996). Each metapopulation is prone to extinction because of its isolation and small size. In this model, the sub-populations are completely independent, and their demographics are not linked (Stith et al., 1996). (3) 'The Mainland-island metapopulation model', where dispersers from the mainland can reach each of the small sub-populations (Ricklefs and Miller, 2000). It is therefore assumed the mainland population does not go extinct (Ricklefs and Miller, 2000). If the mainland population does not go extinct neither does the metapopulation (Hanski and Simberloff, 1997).

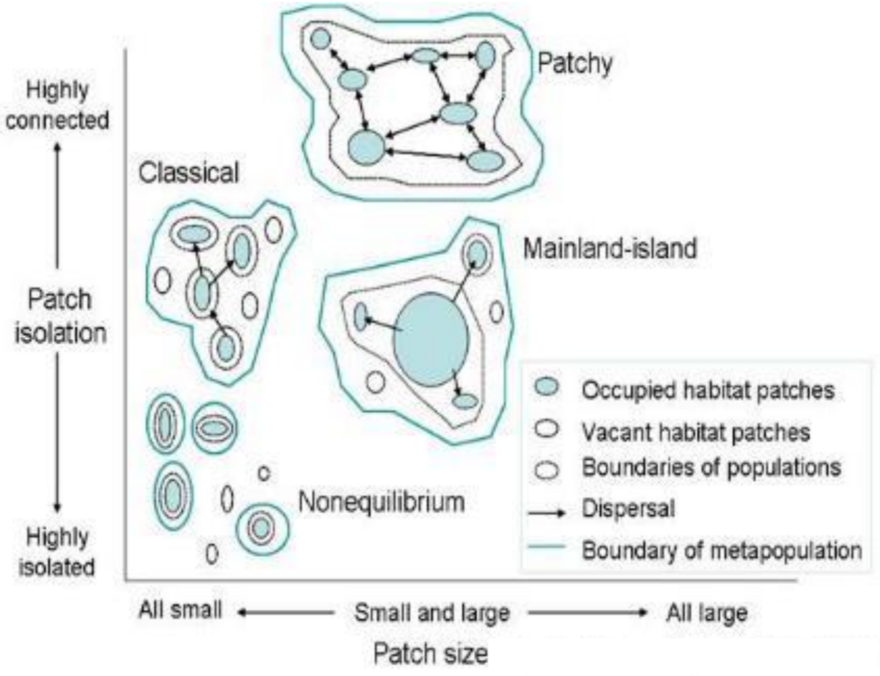


**FIGURE 1.5**. Metapopulation structure (Based on Stith et al., 1996; Harrison and Taylor, 1997).

The sub-populations of skinks within Bermuda represent a combination of the classical metapopulation structure and a mainland-island model (Fig 1.5). All the sites are small, isolated, vary in quality (e.g. in terms of food availability, cover, nest sites, impact of predation), and mostly independent with no migration. However, it is possible that interactions such as movement of individuals from one sub-population to another is possible between a few islands with Castle harbour, particularly Southampton Island, where rock falls or high sea levels wash skinks across to other low lying islands, especially during tropical storms and hurricanes. However, any migration that does occur is severely restricted. For some sub-population's emigration/immigration is not possible because of environmental conditions, the distance between islands or the dispersal ability of the species. As a result of this lack of recolonization, some of the isolated sub-populations may be more susceptible to extinction.

Biology of metapopulations has become a key issue in conservation, especially with respect to how the dynamics, long-term survival and evolution of species are affected by habitat fragmentation (see Hanski, 1994; Hastings and Harrison, 1994; Hanski and Gilpin, 1997; Ricklefs and Miller 2000). Therefore, models of metapopulation dynamics are important tools in the field of conservation biology and island biogeography.

One of the consequences of increased habitat fragmentation is reduced potential for maintaining genetic variation in local populations and across the entire metapopulation (Frankham, 1998). Although genetic differentiation between Bermuda skink sub-populations is not the subject of this thesis, the results are currently being analysed separately to further inform conservation measures.

Recovery plans were put into place in 2010 by the Government of Bermuda to protect the species. They planned to do this by (1) restoring populations of the skink island-wide to reach sustainable levels; (2) ensuring suitable habitat for the skinks; (3) ensuring active reproduction and growth; and (4) reducing fragmentation of the population. However, the plans do not make it clear at what point the population is deemed to reach a sustainable level; how suitable habitat will be maintained; how reproduction and growth of skink populations could be demonstrated at a minimum of 75% of sites deemed suitable; or how fragmentation of the population would be measured and therefore reduced. Consequently, the imprecise objectives stated in the action plan make achieving targets problematical, and as a result there has been no regular monitoring of the skink populations on Bermuda or its offshore islands (Government of Bermuda, 2003) and no research has been published on the Bermuda skinks since 2005 (Edgar et al., 2010).

In 2013, a safety-net population of 12 adult skinks were collected for a captive breeding program at Chester Zoo (UK) that could secure a breeding population in the event of a natural disaster that decimated the wild population. In September 2017 skinks were successfully bred outside of Bermuda for the first time, producing seven hatchlings from two clutches. As a result, husbandry guidelines are currently being complied which could help optimise captive conditions and breeding success, potentially leading to reintroductions back to the wild in the near future.

## 1.7 CURRENT THREATS TO BERMUDA SKINKS

### *1.7.1 Habitat destruction, loss and fragmentation*

Although the skinks were once a common sight across Bermuda historically (Jones et al., 1859; Cope, 1861), no records of skink presence in the northeast of Bermuda, particularly in St. Georges, exist. Although it is possible that the skinks were present but just not recorded, this seems unlikely as St. Georges was the first island to be settled in July 1612 by sixty British settlers (Cook et al., 1980). Since then, extensive methods have been used to burn and clear land for keeping livestock (Lee, 2015) and numerous buildings and fortifications were constructed from the native coral limestone rock, in some cases extracted from coastal areas (Verrill, 1907; Sterrer et al., 2004). During World War 2 (1939 – 1945), many small islands in Castle Harbour were amalgamated or eliminated to provide a land-based airfield for US military aircraft in Bermuda (Forbes, 2019). Bermudians also launched into shipping, a highly successful industry until the advent of steam in the early nineteenth century, taking advantage of the prolific Bermuda cedar for timber (Wilkinson, 1973; Sterrer et al., 2004). Tourism then started to bring money and development in the form of new hotels and growing towns across Bermuda. If skinks did exist in St. Georges originally, their populations would have struggled to survive the continuous anthropogenic disturbances in the area.

By 1965, the mainland skink populations had become noticeably smaller and their distribution more fragmented than the offshore island populations (Wingate, 1965). Over the past century, there has been vast changes in skink habitat across Bermuda, often going from one extreme to the other (e.g. barren to densely vegetated; Fig 1.6). Although Bermuda's agricultural importance has declined, the pressures of development continue to grow due to intense human activity which is perhaps one of the leading causes of the skink's population decline (Wingate, 1998).

**FIGURE 1.6**. Habitat changes: A) Barren coastline of Castle Island in the early 1900s and B) Present day highly vegetated aerial image of Castle Island (*Photo Credit: Verrill, AH. 1902; Bermuda Aerial Media, 2017*).

The country remains one of the most densely populated in the world with 1,160 people per km² (Sterrer, 1998; Government of Bermuda, 2016), in comparison the United Kingdom has 274 people per km² (Office for National Statistics, 2018). The demand for housing has grown with less than 33% of open space land remaining, (including National Parks, Nature Reserves, rural areas, golf courses and other recreational areas. Undisturbed areas of natural vegetation are rapidly diminishing (Government of Bermuda, 2018a). The pressure on biodiversity is intensified further by the arrival of more than half a million tourists each year (Government of Bermuda, 2018b).

Even on Bermuda's mainland there was an estimated 95% – 99% loss of the native Bermuda cedar (*Juniperus bermudiana*) in the 1950s following the accidental introduction of two coccoid scale insects (Gillespie and Clague, 2009). Although subsequent reforestation using a scale-resistant strain has returned the cedar to roughly 10% of its former abundance (Procter and Fleming, 1999), these efforts have been hampered by the introduction of highly invasive fast-growing casuarinas (*Casuarina equisetifolia*), Brazil pepper (*Schinus terebinthifolius*), Indian laurel (*Ficus microcarpa*), asparagus fern (*Asparagus densiflorus*) and other exotics into much of the cedar habitat (Sterrer, 1995). This has caused extensive ecological problems by over-shading native plants and increasing soil erosion (because of extensive underground root systems). Although up to 20% of Bermuda's land area is covered by forest, woodland or scrub habitat (Procter and Fleming, 1999), only very small and fragmented areas of natural habitat remain that are suitable for skinks (Sterrer, 1998; Sterrer et al., 2004) resulting in small pockets of sub-populations, which puts the species at greater risk of extinction.

### *1.7.2 Climate change*

Bermuda is particularly vulnerable to climate change due to a number of factors. For example, the small size of the island limits the options available for adaptation to climate change and sea level rise. Likewise, the relatively low-lying nature of the land and the island's geographical position makes it vulnerable to tropical storm activity and associated storm surge. The impacts of these factors are accentuated by increasing urbanisation and high population density; the close proximity to the coastline of much of the housing and infrastructure; limited natural resources such as fresh water; depletion of the island's non-renewable resources; and the substantial distance to major international markets that drives a dependence on imports including food and oil.

As the sea level rises, the impact of a hurricane on Bermuda's shores will have a significantly greater impact on coastal erosion due to increased storm surge (Stevenson, 2014). A projected 55.8% of Bermuda's beaches and dunes, and 52.7% of the rocky coastal habitat would be lost and more than 1,977 buildings would be affected with a 2 m sea level rise - the predicted maximum sea level rise likely this century (Glasspool, 2008; Pfeffer et al., 2008). Planning can play a key role in tackling climate change by helping to shape developments. Providing green spaces can help to reduce carbon dioxide emissions; minimise energy consumption; minimise vulnerability to flood risk and high temperatures; improve air quality, manage flood and surface water; and positively build community resilience to problems such as extreme heat or flood risk.

Although the majority of the tropical storms and hurricanes that form in the Atlantic bypass Bermuda, when hurricanes do hit Bermuda, usually between September and October (Bermuda Weather Service, 2017), they are quite unpredictable in nature as they change direction and intensity and can get dangerous. However, the damage is often minimal (Bhattacharya, 2019). Notably there have been four hurricanes in the past decade (Hurricane

Fay, Gonzalo, Fabian and Nicole) that have caused significant damage to the island (Bhattacharya, 2019) not only uprooting vegetation, but causing flooding, structural damage to buildings, and the high winds weaken the limestone and cause rock falls, also destroying skink habitat (Fig. 1.7).

**FIGURE 1.7**. Rock fall on Charles Island (before and after) following hurricane Nicole in 2016. (*Photo Credit: Dept. Conservation Services. 2000; Turner, H. 2016*).

### *1.7.3 Pollution*

Litter waste is a serious threat for marine environments and the negative effects of this debris on wildlife are well documented, such as sea turtles ingesting plastics floating in the ocean (Schuyler et al., 2014), whilst marine mammals, birds and fish are often entangled in non-biodegradable materials (Derraik, 2002). Although plastic granules progressively disappear through oxidative ageing and other degradational processes, the amount of oceanic litter is increasing (Thompson et al., 2004; Cole et al., 2011; Bergmann and Klages, 2012) and plastic pollution is a common feature on Bermuda's beaches (Gregory, 1983; Law et al., 2010). Additionally, plastics contain polycyclic aromatic hydrocarbons (PAH), polychlorinated biphenyls (PCBs), bisphenol A (BPAs), phalates, dioxins and dichlorodiphenyltrichloroethane (DDT). These can disrupt the human endocrine system and cause cancers and birth defects (Schug et al., 2011; Maqbool et al., 2016). Equally, plastic waste is capable of absorbing contaminants (Cózar et al., 2015) and can provide reservoirs for mosquitoes, *Aedes aegypti* to breed. These transmit viruses that cause dengue fever (DF), dengue haemorrhagic fever (DHF) and dengue shock syndrome (DSS). As dengue is one of the most widespread infectious diseases globally, it is a major public health problem with almost a half of the global human population at risk (Hoff and Foley, 2011; Arrizabalaga et al., 2016).

Discarded litter washing up on Bermuda’s coastlines also has a major impact on terrestrial species such as the Bermuda skinks. Glass and plastic bottles left in an upright position act as lethal traps for invertebrates such as cockroaches (*Periplaneta americana*) and small crustaceans such as woodlice (*Armadillidium vulgare*) which, in turn, attracts the skinks. As the skinks have clawed feet they cannot escape (unlike anoles) and remain trapped, eventually dying of starvation, drowning or heat stress (Hill, 2003; Fig. 1.8). Indeed, the decline of skinks on Castle and Charles Island may be mostly due to the litter left there as they are subject to high levels of anthropogenic activities during the summer months (Garber, 1988).

**FIGURE 1.8**. The recent remains of a Bermuda skink found in a discarded bottle at Daniels Head. (*Photo Credit: Turner, H. 2015*).

### *1.7.4 Predation and interspecific competition*

Historically, the only native predator of skinks may have been the Bermuda hawk, recently described from fossil records (Olson, 2008). Although it is long extinct, records of raptors observed on the island in 1603 may relate to the hawk. Exact time of extinction is not known, but presumably followed human settlement in the early 17th century and may have been the result of hunting and the introduction of invasive species (Birdlife, 2019). Occasional migratory birds such as egrets may predate skinks (Edgar et al., 2010) as their diet frequently includes small reptiles (Baxter and Fairweather. 1989). However, it is unlikely they had many predators or needed to compete with other species for resources (Edgar et al., 2010). Natives such as giant land crabs (*Cardisoma guanhumi*) and hermit crabs (*Coenobita clypeatus*) occupy coastal areas similar to the skinks and although they are predominately vegetarian they will occasionally scavenge bird nests too (Government of Bermuda, 2019a; Government of Bermuda, 2019b). The only other extant pre-colonial terrestrial vertebrate is the diamondback terrapin (*Malaclemys terrapin*) which occupies brackish coastal tidal marshes and is unlikely to encounter the skinks.

Bermuda is particularly vulnerable to the introduction of invasive species through the importation of food, (i.e. plants, animals, seeds, fruit) or other consumer products which could accidentally introduce an invasive species that will seriously damage the environment (Government of Bermuda, 2018b). Of more than 1,600 resident terrestrial plants and animal species, only 27% are native (Sterrer et al., 2004). Deliberate and accidental introductions may have had a dramatic effect on skink populations (Raine, 1998; Wingate, 1998; Davenport et al., 2001). The first such animals on the island were probably domestic pigs, deliberately put ashore by Spanish sailors in about 1560 so that they could serve as food for passing ships (Sterrer et al, 2004). By 1593, feral pigs were already quite numerous and had destroyed breeding colonies of the Bermuda petrel (and possibly even sea turtles' nests) on the main islands (Lee, 2015). Although there are no records of the feral pigs being removed from the island, any such livestock would have been unlikely to survive the famine of 1615 (Verrill 1902; Sterrer et al., 2004).

The brown rat (*Rattus norvegicus*), the black rat (*Rattus rattus*) and feral and domestic cats (*Felis domesticus*) arrived in shipments to Bermuda sometime before 1614 (Government of Bermuda, 2015). The mouse (*Mus musculus*) was another early (but undocumented) arrival (Sterrer et al., 2004). In the 17th century these abundant introduced species became highly destructive on the islands and were controlled on Bermuda's offshore islands by the repeated burning of the vegetation (Verrill, 1902) and skinks may have been eliminated from many islands as a result. However, thousands of feral and domestic cats remain on Bermuda's mainland and cases of domestic cats catching and killing skinks have been frequently reported (Wingate, 1965; M. Outerbridge, 2016, pers. comm.). For example, in the 1980s an estimated 50 cats became feral on Frick's property at Castle Point (on the mainland) and during this time the owner of the property observed the skink population to decline (Garber, 1988). The cats have since been eliminated and the area restored to native vegetation, but careful observations have failed to find any lizards. Verrill (1902) also suggested rats' prey on skink eggs and young.

Competitors introduced to Bermuda include the marine cane toads, (also known as giant toads, *Rhinella marina* formerly *Bufo marinus*), that were brought into Bermuda in 1885 from Guyana to control pest insects, particularly the cockroaches and centipedes, that are abundant on the island (Dunn and Conant, 1937). The toads compete for terrestrial invertebrate prey and even ingest smaller skinks (Wingate, 2011). Following an explosion in the number of toads on Nonsuch Island after a freshwater pond was created, skinks were observed dying after ingesting toadlets, which contain bufotoxins (Raine, 1998). Since then, a barrier has been erected around the perimeter of this particular pond and it seems to have prevented breeding of toads on Nonsuch Island, but they are common throughout Bermuda (Government of Bermuda, 2018c). A similar incident occurred with another scincid lizard in Australia, the blue tongued skink (*Tiliqua scincoides intermedia*). As soon as invasive cane toads were introduced, skink populations declined as a direct result of ingesting the neurotoxic toads (Price-Rees et al., 2010). Cane toads are therefore a significant threat to the Bermuda skinks (Davenport et al., 2001).

Graham's anole (*Anolis grahami*), native to Jamaica, were initially introduced to Bermuda in 1905 (Wingate, 1965) to control the fruit fly populations that damaged local crops (Losos, 1996). The Antiguan anole (*Anolis bimaculatus leachii*), locally known as 'the Warwick lizard', and the Barbados anole (*Anolis roquet extremus*), known as 'the Somerset lizard', were then accidently introduced in the early 1940s (Losos, 1996). The Jamaican anoles are now the most abundant of the three introduced lizards (Bacon et al., 2006a). As adult Jamaican anoles have been observed consuming juvenile skinks on Castle Island (Griffith and Wingate, 1994) it is likely the larger Antiguan and Barbados anoles would be able to as well (Griffith and Wingate, 1994; Hailey et al., 2011). Now with well-established populations they have the potential to spread island-wide and compete directly with skinks (Stroud et al., 2017).

In 2011 two species of gecko, the Mediterranean or Turkish gecko (*Hemidactylus turcicus*) and the Common or Asian house gecko (*Hemidactylus frenatus*) and the Cuban brown anole (*Anolis sagrei*) were reported to be accidentally introduced to Bermuda from cargo imports (Jones, 2013). Although these species have become established invasives in other parts of the world, currently their populations seem to be restricted to just a few areas on the mainland (Stroud et al., 2017).

All common crows (*Corvus brachyrynchos*) found in Bermuda are descended from one pair that were being kept as pets which escaped in 1848 (Sterrer et al., 2004). Their population in Bermuda has been increasing since the late 1860s and they frequently visit the Castle Harbour islands to consume the white-tailed tropicbird (*Phaethon lepturus catsbyii*) eggs and chicks (Maderios, 2011). The crows have never been directly observed to predate on the Bermuda skinks, but they will eat a wide variety of prey that includes lizards (McGill University, 2008) so it is very possible they could be consuming skinks as well. Similarly, many feral chickens are found throughout the mainland with more than 20,000 estimated in Bermuda (Government of Bermuda, 2017). Chickens are also opportunistic feeders and can even predate adult skinks (Fig. 1.9).

**FIGURE 1.9**. Chicken eating an adult five lined skink *Plestiodon fasciatus* in North America. (*Photo Credit: Natural chicken keeping, 2015*).

Great kiskadee flycatchers (*Pitangus sulphuratus)* were introduced in 1957 from Trinidad as a biocontrol agent for the invasive Jamaican anoles (*Anolis grahami*) (Wingate, 1965), possibly because lizards form a common part of their diet in their natural range (Chadee et al., 1990). However, this initiative was unsuccessful, as both species (Great kiskadees and Graham's anoles) are now widespread across Bermuda (Stroud et al., 2017). As the kiskadees have been observed preying on the skinks, they present a real threat to the skink's survival (Bennett and Hughes, 1959; Samuel, 1975; Raine, 1998; Davenport et al., 2001; Thomas, 2004; Bacon et al. 2006a). Additionally, kiskadees have played a significant role in the population declines of a number of other native species, including the Bermuda land snails (*Poecilozonites circumfirmatus* and *Poecilozonites bermudensi*) and extinction of the Bermuda cicadas (*Tibicen bermudiana*) (Cheeseman and Clubbe, 2007).

When the endemic Bermuda night heron became extinct, a related species - the yellow-crowned night heron (*Nyctanassa violacea*) - was deliberately introduced (1976 – 78) to control the burgeoning land crab population (Kushlan et al., 2011). However, unlike the extinct night heron, the introduced night heron does not have a crab-specific diet and although it did reduce the crab population, it also preyed upon the skinks (Griffith et al., 1991; Griffith and Wingate, 1994; Davenport et al., 1997; Wingate, 1998). Nevertheless, a sample of heron pellets collected in 1997 contained no traces of Bermuda skink material (Davenport et al., 1997). As samples were collected twenty years after the herons were introduced, the skink population may already have been depleted. Equally, the small sample size of pellets collected, the time of the year samples were collected and the fact that night herons are opportunistic feeders (Nellis, 2001), means that impacts on the skinks may still be possible.

There are now a host of introduced or non-native species that have become established on Bermuda that pose a real threat not only to the Bermuda skink, but to the entire ecosystem (Outerbridge et al., 2003) through predation, competition, reducing food web complexity, hybridisation, competitive exclusion, and increasing the risk of extinction of native species (Mooney and Cleland, 2001). To date 1,139 non-native species have been recorded in Bermuda (414 plant species, 699 invertebrate species, 26 vertebrate species) (Glasspool et al., 2003). By contrast with other overseas territories, Bermuda has 175% more non-native species recorded than the second highest St Helena (Table 1.1), This is likely due to a lack of in depth recent studies, particularly in the Caribbean region, as a result these records may significantly under-estimate the true number of non-native species present (Varnham, 2006). It is also likely that a combination of factors including Bermuda's high population, remoteness and optimal environmental conditions that would have allowed this number of non-natives to establish. However, the South Sandwich Islands currently have no non-native species (Varnham, 2006).

**TABLE 1.1**

The number of non-native species recorded per British Overseas Territory in relation to territory size (km²), estimated population size and distance to the mainland (km²).

| British Overseas Territory | Recorded no. of Non-Native Species | Territory Size (km²) | Estimated Population Size (2018 – 2019) | Distance to Mainland (km²) |
|---|---|---|---|---|
| Anguilla | 212 | 91 | 14,869 | 953 |
| Ascension | 167 | 90 | 806 | 3,041 |
| Bermuda | 1,139 | 54 | 62,506 | 1,052 |
| British Antarctic Territory | 5 | 1,709,400 | 0<br>50 non-permanent in winter, over 400 in summer (research personnel) | 5,716 |
| British Indian Ocean Territory | 234 | 60 | 0<br>3,000 non-permanent (UK and US military and staff personnel) | 1,943 |
| British Virgin Islands | 31 | 153 | 31,758 | 551 |
| Cayman Islands | 109 | 264 | 68,076 | 512 |
| Cyprus Sovereign Base Areas | 0 | 255 | 7,700<br>8,000 non-permanent (UK military personnel and their families) | 84 |
| Falkland Islands | 224 | 12,713 | 3,377<br>1,350 non-permanent (UK military personnel) | 520 |
| Gibraltar | 80 | 6.5 | 33,701<br>1,250 non-permanent (UK military personnel) | 0 |
| Guernsey | 11 | 65 | 62,792 | 48 |
| Isle of Man | 25 | 571 | 84,077 | 38 |
| Jersey | 12 | 118 | 106,800 | 22 |
| Montserrat | 39 | 101 | 5,215 | 428 |
| Pitcairn Islands | 46 | 47 | 50<br>6 non-permanent | 3,578 |
| South Georgia | 51 | 3,528 | 0<br>99 non-permanent (officials and research personnel) | 1,799 |
| South Sandwich Islands | 0 | 310 | 0<br>99 non-permanent (officials and research personnel) | 2,354 |
| St Helena | 414 | 121 | 4,349 | 1,878 |
| Tristan de Cunha | 129 | 207 | 300<br>9 non-permanent | 2,775 |
| Turks & Caicos Islands | 22 | 430 | 38,191 | 913 |

The Government of Bermuda's Department of Environment and Natural Resources is continuing to control invasive species. Nevertheless, it has been challenging particularly as some of the key traits identified in invasive species are: (1) high population growth rate; (2) wide climatic or environmental tolerance; (3) short generation time; (4) prolific or consistent reproduction; (5) small seed or egg size; (6) good dispersal; (7) high capacity for uniparental reproduction; (8) absence of specialised germination or hatching requirements; (9) high competitive ability; and (10) ability to escape or survive natural enemies (Mooney and Cleland, 2001; Whitney and Gabler, 2008). Without prior knowledge of invader interactions, removal of only a single invader can lead to an increase in the population size of other invasive species or decrease the population size of native species (Zavaleta, Hobbs and Mooney, 2001).

Invasive species may even affect native species indirectly by changing disease dynamics which can cause catastrophic species loss with knock-on effects for community structure (Prenter et al., 2004). Parasites may play a key role in mediating the impacts of biological invasions. For example, invasive hosts often have fewer parasite species and a lower prevalence of parasites than native hosts, which may provide them with a competitive advantage (Lymbery et al., 2014). Equally, if invasive hosts introduce new parasites, then these may be transmitted to native hosts, leading to the emergence of new disease in the natives (Daszak et al., 2000).

Ranaviral disease in amphibians has been studied intensively during the last decade, as associated mass-mortality events are considered to be a global threat to wild animal populations (Daszak et al., 2000). Since the late 1990s, ranaviruses have also been detected more frequently in reptiles. Cases have been described in various chelonian, snake and lizard species, including brown anoles (*Anolis sagrei*), Asian glass lizards (*Dopasia gracilis*), green anoles (*Anolis carolinensis*), green iguanas (*Iguana iguana*), and a central bearded dragon (*Pogona vitticeps*) (Stöhr et al., 2013). Therefore, invasive anoles could potentially pose a threat to the Bermuda skinks as vectors of disease. Continued monitoring of all invasive species would help to detect early warnings signs (i.e. skin lesions, upper respiratory tract issues, lethargy and anorexia) in lizards.

### *1.7.5 Toxicological threats*

A high incidence of abnormalities observed in local cane toads, suggests they are affected by chemical pollutants, such as pesticides (Bacon et al., 2006b; Linzey et al., 2003). Rodenticides used in Bermuda and widely across the world, contain potent chemicals such as brodifacoum, an anticoagulant poison, and a known toxin to mammals, birds and reptiles (Department of Conservation New Zealand, 2001). Reptiles in particular have been reported to be more sensitive to the effects of persistent toxins in the environment than birds and mammals (Hall, 1980). Although it has been suggested that invertebrates are unlikely to be directly killed by brodifacoum, residues of brodifacoum have been found in beetles collected from bait stations on Stewart Island (Wright and Eason, 1991). Equally, Godfrey (1985) reported secondary poisoning in a zoo where insectivorous birds such as golden plovers (*Pluvialis apricaria*), honey creepers (*Cyanerpes* sp.), finches (Aves: Fringillidae), thrushes (Aves: Turdidae), warblers (Aves: Parulidae), crakes (Aves: Rallidae), and rufous-faced antpittas (*Grallaria erythrotis*) all died due to ingesting ants and cockroaches that had eaten brodifacoum baits. The risks of non-target mortality and contamination after pest control must be carefully balanced against the benefits (Godfrey, 1985). For example, the eradication of rabbits using brodifacoum on Round Island, Mauritius in 1986 illustrates this most clearly. Telfair's skinks and other lizards on the island were considered at risk from poisoning by eating poisoned insects and/or bait and some were killed in the process (Merton, 1987). Three years after eradication of rabbits there was a dramatic regeneration of vegetation and marked increase in the number of lizards including Telfair's skinks (North et al., 1994). The eradication of rats from Korapuki Island in 1986 resulted in a 10-fold increase in lizard numbers in three years and a 30-fold increase in six years (Towns, 1994). This resulted in significantly improved survival rate of lizards with the reduction of introduced predators. However, if poison is used, bait stations should be off the ground to reduce possibilities of soil and water contamination leading to terrestrial invertebrates and even skinks accessing the bait.

## 1.8 THESIS OVERVIEW

Data were collected for this thesis on skinks in Bermuda between April and July 2015 and continued through the same time period in 2016 and 2017. *Chapter 2* involved using passive integrated transponder (PIT) tags for the first time in this species, as a long-term marking method for capture-mark-recapture sampling in two of the largest skink sub-populations. The data were analysed using a robust design model that provided precise estimates of abundance, capture and survival probabilities and temporary emigration. In *Chapter 3* detection and non-detection surveys were undertaken across Bermuda, and dynamic occupancy models were used with the inclusion of covariates as a tool to explicitly account for occupancy, colonisation, extinction and detection. This study provides an unbiased estimate of occupancy that can be used to compare the status of Bermuda skinks across different sites. Consequently, the research addresses important principles of survey methodology such as imperfect detection, that need to be accounted for in monitoring protocols for other reptile species. *Chapter 4* compares the sizes and rates of growth amongst sub-populations and with historical data. This will provide a better understanding of the differentiation between sub-populations and rates of growth over time. Finally, in *Chapter 5* we provided new evidence that human-induced habitat deterioration may affect the body condition of Bermuda skinks and showed that using a body condition index can provide a surrogate measure of fitness. This could be applied to other elusive and endangered reptiles, especially species that occur over a large area, for which exhaustive sampling is unfeasible.

In addition, data collected from this study will assist with the Bermuda skink's future captive breeding management if reintroductions are to take place.

## 1.9 LITERATURE CITED

Aguire, A., and Lutz, P.L. (2004). Marine turtles as sentinels of ecosystem health: is fibropapillomatosis an indicator? *Ecohealth*, 1:275–283.

Atkinson, I.A.E. (1996). Introductions of wildlife as a cause of species extinctions. *Wildlife Biology*, 2:135–141.

Aresco, M.J. (2005). Mitigation measures to reduce highway mortality of turtles and other herpetofauna at a north Florida lake*. Journal of Wildlife Management,* 69:549–560.

Arrizabalaga, A., Gonzalez, L.M., and Torre, I. (2016). Small mammals in discarded bottles: a new world record. *Galemys*, 28:63–65.

Austin, C.C., and Adler, G.H. (1995). Dispersal and speciation of skinks among archipelagos in the tropical Pacific Ocean. *Evolutionary Ecology*, 9:529–541.

Bacon, J.P., Gray, J.A., and Kitson, L. (2006a). Status and conservation of the reptiles and amphibians of the Bermuda Islands. *Applied Herpetology*, 3:323–344.

Bacon J.P., Linzey, D.W., Rogers, R.L., and Fort, D.J. (2006b). Deformities in cane toad (*Bufo marinus*) populations in Bermuda: Part I. Frequencies and distribution of abnormalities. *Applied Herpetology*, 3:39–65.

Baillie, J., and Groombridge, B. (1996). *1996 IUCN red list of threatened animals*. World Conservation Union: Gland, Switzerland.

Barnes, J., and Edy, F. (1987). *Captive reproduction of the Bermuda rock lizard or skink* (*Eumeces longirostris*). Bermuda, Aquarium, Museum and Zoo: Bermuda.

Baxter, G.S., and Fairweather P.G. (1989). Comparison of the diets of nestling cattle egrets and intermediate egrets in the Hunter Valley, New South Wales. *Australian Wildlife Research*, 16:395–404.

Beaupre, S.J., and Douglas, L.E. (2009). "Snakes as indicators and monitors of ecosystem properties". In: Mullin SJ and Seigel RA (Eds.), *Snakes: ecology and conservation*. Cornell University Press: Ithaca, New York, United States. Pp 244–261.

Beheregaray, L.B., and Caccone, A. (2007). Cryptic biodiversity in a changing world. *Journal of Biology,* 6:9.

Bergmann, M., and Klages, M. (2012). Increase of litter at the Arctic deep-sea observatory Hausgarten. *Marine Pollution Bulletin*, 64:2734–2741.

Bennett, F., and Hughes, L. (1959). Biological control of insect pests in Bermuda. *Bulletin of Entomological Research*, 50:423–436.

Bhattacharya, R. (2019). *Bermuda hurricane*. [Online]. Available at: https://www.bermuda-attractions.com/bermuda_00014c.htm [Assessed 6 May 2019]

BirdLife International. (2019). IUCN red list for birds: species factsheet: *Bermuteo avivorus*. [Online]. Available at: http://www.birdlife.org [Assessed 26 May 2019].

Bolten, A., and Witherington, B. (2003). *Loggerhead sea turtles*. Smithsonian Books: Washington DC, United States.

Bouché, P., Mange, R.N.M., Tankalet, F., Zowoya, F., Lejeune, P., and Vermeulen C. (2012). Game over! wildlife collapse in northern Central African Republic. *Environmental Monitoring and Assessment*, 184:7001–7011.

Brandley, M.C., Wang, Y., Guo, X., Montes de Oca, A.N., Feria-Ortiz, M., Hikida, T., and Ota, H. (2010). Bermuda as an evolutionary life raft for an ancient lineage of endangered lizards. *PLoS ONE,* 5: e11375.

Brandley, M.C., Ota, H., Hikida, T., Montes de Oca, A.N., Feria-Ortiz, M., Guo, X., and Wang, Y. (2012). The phylogenetic systematics of blue-tailed skinks (*Plestiodon*) and the family Scincidae. *Zoological Journal of Linnean Society,* 165:163–189.

Brashares, J.S., Arcese, P., and Sam, K. (2001). Human demography and reserve size predict wildlife extinction in West Africa. *Proceedings of the Royal Society of London B*, 268:2473–2478.

Brewer, M.S. (2014). *Shakers and movers: Earthquakes and tsunamis felt in Bermuda*. Published by Monkey Cup Press, Warwick PG 01 Bermuda.

Bryant, P.J. (2002). *Biodiversity and conservation. Chapter 12: Islands*. School of Biological Sciences, University of California: Irvine, California, United States.

Bugoni, L., Krause, L., and Petry, M.V. (2001). Marine debris and human impacts on sea turtles in southern Brazil. *Marine Pollution Bulletin*, 42:1330–1334.

Burger, J., and Gibbons, J.W. (1998). Trace elements in egg contents and eggshells of slider turtles (*Trachemys scripta*) from the Savannah river site. *Archives of Environmental Contamination and Toxicology,* 34:382–386.

Burns, C.E., Johnston, K.M., and Schmitz, O.J. (2003). Global climate change and mammalian species diversity in U.S. national parks. *PNAS,* 100; 11474–11477.

Case T.J., Bolger D.T., and Richman A.D. (1992). “Reptilian extinctions: The last ten thousand years”. In: Fiedler PL and Jain SK (Eds.), *Conservation Biology*. Springer: Boston, United States. Pp 91–125.

Cash, W.B. (2015). Skink. In: *Access Science.* McGraw-Hill Education. [Online] Available at: https://doi.org/10.1036/1097-8542.627500 [Assessed 14 February 2019]

Ceballos, G., Ehrlich, P.R., and 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*, 2017:04949.

Chadee, D.D., Ganesh, R., and Persad, R.C. (1990). Feeding behaviour of the great kiskadee, *Pitangus suiphuratus*, on fish in Trinidad, West Indies. Living World Journal of The Trinidad & Tobago Field Naturalists' Club 1990-1991. Pp 42–43.

Cheesman, O., and Clubbe, C. (2007). *Dealing with alien invasive species – introduction, overview and conclusions.* Jersey, biodiversity that matters: a conference on conservation in UK overseas territories and other small island communities. Pp. 193–200.

Cheke , A.S., and Hume , J.P. (2008). *Lost land of the dodo: An ecological history of Mauritius, Réunion and Rodrigues*. T & AD Poyser: London, United Kingdom.

Clout, M.N., and Veitch, C.R. (2002). *Turning the tide of biological invasion: the potential for eradicating invasive species*. IUCN SSC Invasive Species Specialist Group: Gland and Cambridge.

Cole, M., Lindeque, P., Halsband, C., and Galloway, T.S. (2011). Microplastics as contaminants in the marine environment: a review. *Marine Pollution Bulletin*, 62:2588–2597.

Cook, C., and Wroughton, J. (1980). “Overseas trade and the colonies”. In: Cook C and Wroughton J (Eds.), *English historical facts 1603–1688. Palgrave historical and political facts.* Palgrave Macmillan: London, United Kingdom. Pp. 165–181.

Cooper J.E. (1984). “Physical Influences”. In: Hoff GL., Frye FL and Jacobson ER (Eds.), *Diseases of amphibians and reptiles*. Springer: Boston, Massachusetts, United States. Pp 607–624.

Cooper, T., and Viernes, K. (2011). Mycophagy in the larger bodied skinks of the genera *Tiliqua* and *Egernia*: are there implications for ecosystem health? *Australian Zoologist*, 35:681–684.

Cooper, W.E., and Vitt, L.J. (1993). Female mate choice of large broad-headed skinks.. *Animal Behaviour,* 45:683–693.

Cope, E.D. (1861). On the reptillia of Sombrero and Bermuda. *Proceedings of the Academy of Natural Sciences of Philadelphia,* 13:312–314.

Copeland, A. (2011). *UK overseas territories and crown dependencies: 2011 biodiversity snapshot*. [Online]. Available at: http://jncc.defra.gov.uk/PDF/ot_biodiversity2011_bermuda.pdf [Assessed 16 September 2018].

Conyers, J., and Wingate, D. (1996). *Plestiodon longirostris. The IUCN red list of threatened species 1996.* [Online] Available at: http://dx.doi.org/10.2305/IUCN.UK.1996.RLTS.T8218A12900393.en [Assessed 23 August 2017].

Coughlan, J.P., Kitson, L., Dillane, E., Davenport, J., and Cross, T.F. (2004). Characterization of six microsatellite loci in the Bermuda skink (*Eumeces longirostris*). *Molecular Ecology Resources,* 4:678–679.

Courchamp, F., Hoffmann, B.D., Russell, J.C., Leclerc, C., and Bellard, C. (2014). Climate change, sea-level rise, and conservation: keeping island biodiversity afloat. *Trends in Ecology and Evolution*, 29:127–130.

Cózar, A., Sanz-Martín, M., Martí, E., González-Gordillo, J.I., Ubeda, B., Gálvez, J.A., Irigoien, X., Duarte, C.M. (2015). Plastic accumulation in the Mediterranean Sea. *PLoS ONE*, 10: e0121762.

Darwin, C.R. (1859). *On the origin of species by means of natural selection, or the preservation of favoured species in the struggle for life*. John Murray: London.

Darwin, C.R. (1909). *The voyage of the beagle*. P.F. Collier & Son: New York, United States.

Daszak, P., Cunningham, A.A., and Hyatt, A.D. (2000). Emerging infectious diseases of wildlife— threats to biodiversity and human health. *Science,* 287:443.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). *A study of populations of the Bermudian rock lizard (skink), Eumeces longirostris, cope* (1861), *on the islands of Nonsuch and Southampton, Bermuda*. Department of Agriculture and Fisheries: Bermuda.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (2001). Threats to the critically endangered endemic Bermudian skink *Eumeces longirostris*. *Oryx,* 35:332–339.

Department of Conservation New Zealand. (2001). *Vertebrate pesticide toxicology manual (poisons).* Department of Conservation New Zealand: Wellington, New Zealand.

Derraik, J.G.B. (2002). The pollution of the marine environment by plastic debris: a review. *Marine Pollution Bulletin*, 44:842–852.

Dobson, A., and Lyles, A. (2000). Black-footed ferret recovery. *Science*, 288:985–988.

Dunn, E.R., and Conant, R. (1937). The herpetological fauna of Bermuda. *Herpetologica,* 1:78–80.

Dutton, D.L., Dutton, P.H., Chaloupka, M., and Boulon, R.H. (2005). Increase of a Caribbean leatherback turtle *Dermochelys coriacea* nesting population linked to long-term nest protection. *Biological Conservation*, 126:186–194.

Edgar, P., Kitson, L., and Glasspool, A. (2010). *Recovery plan for the Bermuda skink, Eumeces longirostris*. Department of Environment and Natural Resources: Bermuda.

Ellison, J.C. (1996). Global ecology and biogeography letters. *Pollen Evidence of Late Holocene Mangrove Development in Bermuda,* 5:315–326.

Emerson, B.C. (2008). Speciation on islands: what are we learning? *Biological Journal of the Linnean Society,* 95:47–52.

Fitch, H. (1956). Life history and ecology of the five-lined skink, *Eumeces fasciatus. University of Kansas Museum of Natural History Publication,* 8:1–156.

Forbes, K.A. (2019). *Airport facts*. [Online]. Available at: http://www.bermuda-online.org/airport.htm [Assessed 20 May 2019].

Frankham, R. (1998). Inbreeding and extinction: island populations. *Conservation Biology,* 12:665–675.

Garber, S. (1988). *Behaviour and ecology of the endangered endemic Bermuda rock lizard (Eumeces longirostris).* Rutgers University: New Jersey, United States.

Gardner, T.A., Barlow, J., and Peres, C.A. (2007). Paradox, presumption and pitfalls in conservation biology: The importance of habitat change for amphibians and reptiles. *Biological Conservation*, 138:166–179.

Gibbons, J.W., and Stangel. P.W. (1999). "Conserving amphibians and reptiles in the new millennium". *Proceedings of the partners in amphibian and reptile conservation (PARC) conference*; 2–4 June 1999. Savannah River Ecology Laboratory: Atlanta (GA), Aiken (SC). Herp Outreach Publication No. 2.

Gibbons, J.W., Scott, D.E., Ryan, T.J., Buhlmann, K.A., Tuberville, T.D., Metts, B.S., Greene, J.L., Mills, T., Leiden, Y., Poppy, S., and Winnie, C.T. (2000). The global decline of reptiles, déjà vu amphibians. *BioScience*, 50:653–666.

Gillespie, R.G., Claridge, E.M., and Goodacre, S.L. (2008). Biogeography of the fauna of French Polynesia: diversification within and between a series of hot spot archipelagos. *Philosophical Transactions of the Royal Society B-Biological Sciences*, 363:3335–3346.

Gillespie, R.G., and Clague, D.A. (2009). *Encyclopedia of islands.* University of California Press: California, United States.

Glasspool, A. (2000). *Skink update.* Bermuda Aquarium, Museum and Zoo: Bermuda.

Glasspool, A., Ward, J., de Silva, H., Sterrer, W., and Furbert, J. (2003). “A biodiversity strategy and action plan for Bermuda – a recipe for success”. In: Pienkowski M (Ed.), *A sense of direction: a conference on conservation in UK overseas territories and other small island communities*. UK Overseas Territories Conservation Forum: Over Norton, United Kingdom. Pp. 34–38.

Glasspool, A., and Outerbridge, M. (2004). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island, Castle Harbour.* Bermuda Zoological Society: Bermuda.

Glasspool, A. (2008). *The impact of climate change on Bermuda*. Bermuda National Trust: Bermuda.

Godfrey, M.E. (1985). Non-target and secondary poisoning hazards of “second generation” anticoagulants. *Acta Zoologica Fennica,* 173:209–212.

Government of Bermuda. (2003). *Bermuda biodiversity strategy and action plan.* Department of Environment and Natural Resources: Bermuda.

Government of Bermuda. (2015). *Rats.* Department of Environment and Natural Resources. [Online] Available at: http://environment.bm/rats/ [Accessed 27 September 2016].

Government of Bermuda. (2016). *2016 Population and housing census report*. Department of Statistics. [Online]. Available at: https://www.gov.bm/sites/default/files/2016%20Census%20Report.pdf [Assessed 16 September 2018].

Government of Bermuda. (2017). *Feral chickens.* Department of Environment and Natural Resources. [Online] Available at: https://environment.bm/feral-chickens/ [Accessed 10 September 2018].

Government of Bermuda. (2018a). *The draft Bermuda plan 2018. Review and strategy report*. Department of Planning: Bermuda. Pp 1–68.

Government of Bermuda. (2018b). *Facts and figures 2018*. Department of Statistics: Bermuda. [Online]. Available at: https://www.gov.bm/sites/default/files/2018-Facts-Figures_0.pdf [Assessed 16 April 2019].

Government of Bermuda. (2018c). *Marine or giant toad* (*Bufo marinus*). Department of Environment and Natural Resources. [Online]. Available at: https://environment.bm/toads [Assessed 16 April 2019].

Government of Bermuda. (2019a). Giant land crab (*Cardisoma guanhumi*). Department of Environment and Natural Resources. [Online]. Available at: https://environment.bm/giant-land-crab [Assessed 18 June 2019].

Government of Bermuda. (2019b). *Land hermit* crab (*Coenobita clypeatus*). Department of Environment and Natural Resources. [Online]. Available at: https://environment.bm/land-hermit-crab [Assessed 18 June 2019].

Gregory, M.R. (1983). Virgin plastic granules on some beaches of Eastern Canada and Bermuda. *Marine Environmental Research*, 10:73–92.

Griffith, H., Wingate, D.B., and deLaey, R. (1991). *Introduced birds as probable agents of population changes.* Bermuda Aquarium, Museum and Zoo: Bermuda.

Griffith, H., and Wingate, D.B. (1994). *Eumeces longirostris* (Bermuda rock lizard or skink). Predation. *Herpetological Review,* 25:26.

Hailey, A., Wilson, B., and Horrocks, J. (2011). *Carribean island herpetofaunas: conservation biology and the wider carribean.* Volume 1. Brill Publishing: The Netherlands.

Hall, R.J. (1980). *Effects of environmental contaminants on reptiles: a review.* United States Department of the Interior, Fish and Wildlife Service: Washington D.C, United States. Special Scientific Report - Wildlife No. 228.

Hammond, M.P. (2000). *An analysis of Bermuda Skink (Eucemes longirostris) populations on Castle Island, Bermuda.* Bermuda Aquarium, Museum and Zoo: Bermuda.

Hanski, I. (1994). A practical model of metapopulation dynamics. *Journal of Animal Ecology*, 63:151–162.

Hanski, I., and Gilpin, M.E. (1997). *Metapopulation biology: Ecology, genetics, and evolution*. Academic Press: San Diego, California, United States.

Hanski, I., and Simberloff, D. (1997). “The metapopulation approach: its history, conceptual domain, and application to conservation”. In: Hanski IA and Gilpin ME (Eds.), *Metapopulation biology: ecology, genetics, and evolution*. Academic Press: San Diego, California, USA. Pp 5–26.

Harrison, S., and Taylor A.D. (1997). Empirical evidence for metapopulation dynamics. In: Hanski IA and Gilpin ME (Eds.). *Metapopulation biology: ecology, genetics, and evolution*. Academic Press: San Diego, California, United States. Pp 27-42.

Hartup, B.K. (1996). Rehabilitation of native reptiles and amphibians in DuPage county, Illinois. *Journal of Wildlife Diseases*, 32:109–112.

Hastings, A., and Harrison, S. (1994). Metapopulation dynamics and genetics. *Annual Review of Ecology and Systematics*, 25:167–188.

Heatwole, H. (1976). *Reptile ecology*. University of Queensland Press: Australia.

Hedrick, P.W. (2009). Conservation genetics and North American bison (*Bison bison*). *Journal of Heredity*, 100:411–420.

Hedges, S.B., and Conn, C.E (2012). A new skink fauna from Caribbean islands (Squamata, Mabuyidae, *Mabuyinae*). *Zootaxa*, 3288:1–244.

Heilprin, A. (1889). *The Bermuda islands- a contribution to the physical history and zoology of the Somers archipelago*. Heilprin: Philadelphia, United States.

Hill, R. (2003). *Biodiversity project sheds light on island skinks*. The Royal Gazette: Bermuda.

Hoff, M.J., and Foley, J.A. (2011). Global-scale relationships between climate and the dengue fever vector, *Aedes aegypti*. *Climate Change*, 48:448–463.

Hoffmann, M., Hilton-Taylor, C., Angulo, A., Bohm, M., *et al.* (2011). The impact of conservation on the status of the world's vertebrates. *Science*, 330:1503–1509.

Houghton, J. (1997). *Global warming*. Cambridge University Press: Cambridge, United Kingdom.

Huber, M. (2009). Climate change: snakes tell a torrid tale. *Nature*, 457:669–671.

Hughes, L. (2000). Biological consequences of global warming: is the signal already apparent*? Trends in Ecology and Evolution*, 15:56–61.

IUCN. (2015). *Plestiodon longirostris*. [Online]. Available at: http://www.iucnredlist.org/details/8218/0 [Accessed 29 June 2015].

IUCN. (2018a). *IUCN red list for endangered species - Reptiles.* [Online]. Available at: http://www.iucnredlist.org/ [Accessed 1 July 2018].

IUCN. (2018b) *Island ecosystems*. [Online] Available at: https://www.iucn.org/commissions/commission-ecosystem-management/our-work/cems-specialist-groups/island-ecosystems [Assessed 30 July 2018].

IUCN. (2018c). *IUCN - Invasive species*. [Online]. Available at: https://www.iucn.org/theme/species/our-work/invasive-species [Assessed 12 September 2018].

Jones, J., Hurdis, J., and Wedderburn, J. (1859). *The naturalist in Bermuda; a sketch of the geology, zoology and botany of that remarkable group of islands; together with meteorological observations*. Reeves & Turner: London, United Kingdom.

Jones, C.G., Heck, W., Lewis, R.E., Mungroo, Y., Slade, G., and Cade, T. (1994). The restoration of the Mauritius kestrel *Falco punctatus* population. *Ibis*, 137:173–180.

Jones, S. (2013). *Stowaway lizards could threaten our skink*. [Online]. Available at: http://bermudasun.bm/Content/NEWS/News/Article/Stowaway-lizards-could-threaten-our- skink/24/270/70720 [Assessed 01 September 2017].

Kenney. J.S., Smith, J.D.L., Starfield, A.M., and McDougal, C.W. (1995). The long-term effects of tiger poaching on population viability. *Conservation Biology*, 9:1127-1133.

Kier, G., Kreft, H., Lee, T.M., Jetz, W., Ibisch, P.L., Nowicki, C., Mutke, J., and Barthlott, W. (2009). A global assessment of endemism and species richness across island and mainland regions. *Proceedings of the National Academy of Sciences*, 106:9322–9327.

Kierulff, M.C.M., Ruiz-Miranda, C.R., Procopio de Oliveira, P., Beck, B.B., Martins, A., Dietz, J.M., Rambaldi, D.M., and Baker, A.J. (2012). The golden lion tamarin *Leontopithecus rosalia*: a conservation success story. *International Zoo Yearbook*, 46:36–45.

Kingsolver, J.G. (2009). The well-temperatured biologist. *The American Naturalist*, 174:755–768.

Koch, V., Nichols, W.J., Peckham, H., and, Toba, V. (2006). Estimates of sea turtle mortality from poaching and bycatch in Bahia Magdalena, Baja California Sur, Mexico. *Biological Conservation,* 128:327–334.

Kotas, J.E., dos Santos, S., de Azevedo, V.G., Gallo, B.M., and Barata, P.C. (2004). Incidental capture of loggerhead (*Caretta caretta*) and leatherback (*Dermochelys coriacea*) sea turtles by the pelagic longline fishery off southern Brazil. *Fishery Bulletin*, 102:393–399.

Kuschel, G. (1975). *Biogeography and ecology in New Zealand*. Springer: The Netherlands.

Kushlan, J.A., Wingate, D.B., and Madeiros, J.L. (2011). Conservation of the green heron (*Butorides virescens*) on Bermuda. *Journal of Heron Biology and Conservation*, 1:4.

Law, K.L., Morét, S., Maximenko, N., Proskurowski, G., Peacock, E., Hafner, J., and Reddy, C.M. (2010). Plastic accumulation in the North Atlantic subtropical gyre. *Science*, 329:1185–1188.

Lee, D.S. (2015). Gulf stream chronicles: A naturalist explores life in an ocean river. University North Carolina Press: United States.

Levins, R. (1969). Some demographic and genetic consequences of environmental heterogeneity for biological control. *Bulletin of the Entomological Society of America,* 15:237–240.

Linzey, D., Burroughs, J., Hudson, L., Marini, M., Robertson, J., Bacon, J., Nagarkatti, M., and Nagarkatti, P. (2003). Role of environmental pollutants on immune functions, parasitic infections and limb malformations in marine toads and whistling frogs from Bermuda. *International Journal of Environmental Health Research*, 13:125–148.

Losos, J. (1996). Dynamics of range expansion by three introduced species of *Anolis* lizards on Bermuda. *Journal of Herpetology*, 30:204–210.

Lymbery, A.J., Morine, M., Kanani, H.G., Beatty, S.J., and Morgan, D.L. (2014). Co-invaders: The effects of alien parasites on native hosts. *International Journal for Parasitology*: *Parasites and Wildlife*, 3:171–177.

Maas, P.H.J. (2017). Globally extinct reptiles. In: *The Sixth Extinction*. [Online]. Available at: http://www.petermaas.nl/extinct [Assessed 11 September 2018].

Maderios, J. (2011). Breeding success and status of Bermuda's longtail population (white-tailed tropicbird) *Phaethon lepturus catsbyii* at ten locations on Bermuda 2009 – 2011. Department of Conservation Services: Bermuda.

Marcovaldi, A., and Chaloupka, M. (2007). Conservation status of the loggerhead sea turtle in Brazil: an encouraging outlook Maria. *Endangered Species Research*, 3:133–143.

Markham, A. (1996). Potential impacts of climate change on ecosystems: a review of implications for policymakers and conservation biologists. *Climate Research*, 6:179–191.

MacArthur, R.H., and Wilson, E.O. (2001). *The theory of island biogeography*. Princeton University Press: United States.

Maqbool, F., Mostafalou, S., Bahadar, H., Abdollahi, M. (2016). Review of endocrine disorders associated with environmental toxicants and possible involved mechanisms. *Life Sciences,* 145:265–273.

McCarty, J.P. (2001). Ecological consequences of climate change. *Conservation Biology,* 15: 320–331.

McGill University. (2008). *Urban nature information service: ravens and crows.* [Online]. Available at: http://unis.mcgill.ca/en/uw/birds/ravens_crows.html [Accessed 11 August 2016].

Merton, D. (1987). Eradication of rabbits from Round island, Mauritius: a conservation success story. *Dodo, Journal of the Jersey Wildlife Preservation Trust,* 24:19–44.

Mittermeier, R.A., Carr, J.L. Swingland, I.R. Werner. T.B., and Mast. R.B. (1992). "Conservation of amphibians and reptiles". In: Adler K (Ed.), *Herpetology: contemporary research on the biology of amphibians and reptiles.* Society for the Study of Amphibians and Reptiles: Ohio, United States. Pp 59–80.

Mooney, H., and Cleland, E. (2001). The evolutionary impact of invasive species. *PNAS,* 98:5446–5451.

Morton, M.C. (2013). *Setting sail on unknown seas: the past, present and future of species rafting.* [Online] Available at: http://www.earthmagazine.org/article/setting-sail-unknown-seas-past-present-and-future-species-rafting [Accessed 25 August 2015].

Office for National Statistics, (2018). *Estimates of the population for the UK, England and Wales, Scotland and Northern Ireland*. [Online]. Available at: https://www.ons.gov.uk/peoplepopulationandcommunity/populationandmigration/populationestimates/datasets/populationestimatesforukenglandandwalesscotlandandnorthernireland [Assessed 7 March 2019].

Olson, Storrs L. and Meylan, Peter A. (2009). A second specimen of the Pleistocene Bermuda tortoise, *Hesperotestudo bermudae* Meylan and Sterrer. *Chelonian Conservation and Biology,* 8: 211–212.

Nellis, D. (2001). *Common coastal birds of Florida and the Caribbean.* Pineapple Press Inc: United States.

Nielsen, R. (2006). *The little green handbook: seven trends shaping the future of our planet*. Picador: New York, United States.

North, S.G., Bullock, D.J., and Dulloo, M.E. (1994). Changes in vegetation and reptile populations on Round Island Maritius, following eradication of rabbits. *Biological Conservation,* 67:21–28.

Olsson, M. (1994). Nuptial coloration in the sand lizard, *Lacerta agilis*. an intra-sexually selected cue to fighting ability. *Animal Behaviour,* 48:607–613.

Olson, S.L., and Wingate, D.B. (2000). Two new flightless rails (Aves: Rallidae) from the Middle Pleistocene “crane fauna” of Bermuda. *Proceedings of the Biological Society of Washington*. 113:356–368.

Olson, S.L. (2008). A new genus and species of buteonine hawk from quaternary deposits in Bermuda (Aves: Accipitridae). *Proceedings of the Biological Society of Washington*. 121:130–141.

Olson, S.L. (2012). A new species of small owl of the genus *Aegolius* (Aves: Strigidae) from Quaternary deposits on Bermuda. *Proceedings of the Biological Society of Washington*. 125:97–105.

Olsen, S.L., Hearty, P.J., and Pregill, G.K. (2006). Geological constraints on evolution and survival in endemic reptiles on Bermuda. *Journal of Herpetology,* 40:394–398.

Outerbridge, A., Dobson, A., Carey, W. and Ward, J. (2003). “Introduction to Bermuda’s environment and conservation issues”. In: Pienkowski M (Ed.), *A sense of direction: a conference on conservation in UK overseas territories and 18 non-native species in UK overseas territories: a review other small island communities*. UK Overseas Territories Conservation Forum: Over Norton, United Kingdom. Pp 19–31.

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

Pfeffer, W.T., Harper, J.T., and O'Neel, S. (2008). Kinematic constraints on glacier contributions to 21st century sea-level rise. *Science*, 321:1340–1343.

Pincheira-Donoso, D., Bauer, A.M., Meiri, S., and Uetz, P. (2013). Global taxonomic diveristy of living reptiles. *PLoS ONE,* 8: e59741.

Prenter, J., MacNeil, C., Dick, J.T.A., and Dunn, A.M. (2004). Roles of parasites in animal invasions. *Trends in Ecology and Evolution*, 19:385–390.

Price-Rees, S.J., Webb, J.K., and Shine, R. (2010). Predation on toxic cane toads (*Bufo marinus*) may imperil bluetongue lizards (*Tiliqua scincoides intermedia*, Scincidae) in tropical Australia. *Wildlife Research,* 37:166–173.

Pritchard, P.C.H. (1980). The conservation of sea turtles: practices and problems. *American Zoologist*, 20:609–617.

Procter, D., and Fleming, L.V. (1999). *Biodiversity: the UK overseas territories.* Joint Nature Conservation Committee: Peterborough, United Kingdom.

Purvis, A., Gittleman, J.L., Cowlishaw, G., and Mace, G.M. (2000). Predicting extinction risk in declining species. *Proceedings of the Royal Society of London B: Biological Sciences*, 267:1947–1952.

Pyron, R.A., and Camp, C.D. (2007). Courtship and mating behaviours of two syntopic species of skink (*Plestiodon anthracinus* and *Plestiodon fasciatus*). *Amphibia-Reptilia,* 28:263–268.

Raine, A. (1998). *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the critically endangered Bermuda Rock Lizard (Eumeces longirostris).* MSc Thesis. University College London: London, United Kingdom.

Ricklefs, R. E., and G. L. Miller. (2000). Ecology. 4th edition. W.H. Freeman & Co: New York, United States.

Rodrigues, A.S.L. (2006). Are global conservation efforts successful? *Science*, 25:1051–1052.

Rowe, M.P. (1998). *An explanation of the geology of Bermuda*. Bermuda Government Publication. 30 pp.

Rueger, B.F., and von Wallmenich, T.N. (1996). Human impact on the forests of Bermuda: the decline of endemic cedar and palmetto since 1609, recorded in the Holocene pollen record of Devonshire marsh. *Journal of Paleolimnology*, 16:59–66.

Salvador, A., Veiga, J.P., Martin, J., Lopez, P., Abelenda, M., and Puerta M. (2013). The cost of producing a sexual signal: testosterone increases the susceptability of male lizards to ectoparasitic infestation. *Behavioral Ecology,* 7:145–150.

Samuel, D. (1975). *Feeding habits of the kiskadee flycatcher*. Bermuda Biological Station: Bermuda.

Schug, T.T., Janesick, A., Blumberg, B., and Heindel, J.J. (2011). Endocrine disrupting chemicals and disease susceptibility. *The Journal of Steroid Biochemistry and Molecular Biology*, 127:204–215.

Schuyler, Q., Hardesty, B.D., Wilcox, C., and Townsend, K. (2014). Global analysis of anthropogenic debris ingestion by sea turtles. *Conservation Biology*, 28:129–139.

Smith, H.M. (2005). *Plestiodon*: a replacement name for most members of the genus *Eumeces* in North America. *Journal of Kansas Herpetology,* 14:15–16.

Smith, K.F., Sax, D.F., and Lafferty, K.D. (2006). Evidence for the role of infectious disease in species extinction and endangerment. *Conservation Biology*, 20:1349–1357.

Smith, M.J., Cogger, H., Tiernan, B., Maple, D., Boland, C., Napier, F., Detto, T., and Smith, P. (2012). An oceanic island reptile community under threat: the decline of reptiles on Christmas island, Indian ocean. *Herpetological Conservation and Biology*, 7:206–218.

Spotila, J.R., Dunham, A.E., Leslie, A.J., Steyermark, A.C., Plotkin, P.T., and Paladino, F.V. (1996). Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? *Chelonian Conservation and Biology*, 2:209–222.

Soto, J.M.R., Serafini, T.Z., and Celini, A.A.O. (2003). "Beach strandings of sea turtles in the state of Rio Grande do Sul: an indicator of gillnet interaction along the southern Brazilian coast". In: Seminoff JA (Ed.), *Proceedings of the 22nd annual symposium sea turtle biology and conservation*. NOAA Tech Memo NMFS-SEFSC-503. Pp 276.

Sterrer, W. (1995). "Bermuda's natural history in a nutshell". In: *Proceedings of the 1st symposium of fauna and flora of the Atlantic islands*. Boletim do Museo Municipal do Funchal: Funchal, Portugal. Pp 4.

Sterrer, W. (1998). How many species are there in Bermuda? *Bulletin of Marine Science,* 62: 809–840.

Sterrer, W., Glasspool, A., De Silva, H., and Furbert, J. (2004). "Bermuda—An island biodiversity transported." In: Davenport J & Davenport JL (Eds.), *The effects of human transport on ecosystems: cars and planes, boats and trains*. Royal Irish Academy: Dublin. Pp 118–170.

Stevenson, C. (2014). *Assessing the likely impact of rising sea levels on Bermuda*. The Royal Gazette: Bermuda.

Stith, B.M., Fitzpatrick, J.W., Woolfenden, G.E., and Pranty, B. (1996). "Classification and conservation of metapopulations: A case study of the Florida scrub jay". In: McCullough DR (Ed.), *Metapopulations and wildlife conservation*. Island Press, Washington, D.C., United States. Pp 187–215.

Stöhr, A.C., Blahak, S., Heckers, K.O., Wiechert, J., Behncke, H., Mathes, K., Günther, P., Zwart, P., Ball, I., Rüschoff, B., and Marschang, R.E. (2013). Ranavirus infections associated with skin lesions in lizards. *Veterinary Research*, 44:84.

Stroud, J.T., Giery, S.T., and Outerbridge, M.E. (2017). Establishment of *Anolis sagrei* on Bermuda represents a novel ecological threat to critically endangered Bermuda skinks (*Plestiodon longirostris*). *Biological Invasions*, 19:1723–1731.

Tapilatu, R.F., Dutton, P.H., Tiwari, M., Wibbels, T., Ferdinandus, H.V., Iwanggin, W.G., and Nugroho, B.H. (2013). Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: a globally important sea turtle population. *Ecosphere*, 4:25.

Tinkle, D.W. (1979). Long-term field studies. *BioScience*, 29:717.

Thomas, M. (2004). *The natural history of Bermuda.* Bermuda Zoological Society: Bermuda.

Thompson, R.C., Olsen, Y., Mitchell, R.P., Davis, A., Rowland, S.J., John, A.W.G., McGonigle, D., and Russell, A.E. (2004). Lost at sea: where is all the plastic? *Science*, 304:838.

Todd, B.D., Willson, J.D., and Gibbons, J.W. (2010). "The global status of reptiles and causes of their decline". In: Sparling DW., Bishop CA and Krest S (Eds.), *Ecotoxicology of amphibians and reptiles*. Second Edition. CRC Press: Pensacola, Florida, United States. Pp 47–67.

Tomillo, P.S., Saba, V.S., Piedra, R., Paladino, F.V., and Spotila, J.R. (2008). Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. *Conservation Biology*, 22:1216–1224.

Towns, D.R. (1994). The role of ecological restoratation in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *New Zealand Journal of Zoology,* 20:457–471.

Uetz, P., Freed, P., and Hošek. J. (2018). The reptile database. [Online] Available at: http://www.reptile-database.org [Accessed 14 February 2018].

Valencia-Aguilar, A., Cortes-Gomez, A.M., and Ruiz-Agudelo, C.A. (2013). Ecosystem services provided by amphibians and reptiles in Neotropical ecosystems. *International Journal of Biodiversity Science, Ecosystem Services and Management*, 9:257–272.

Van der Ree, R., Smith, D.J., and Grilo, C. (2015). "Handbook of road ecology". In: Andrews KM., Langen TA and Struijk RPJH (Eds.), *Reptiles: overlooked but often at risk from roads*. Wiley-Blackwell: United Kingdom. Pp 271–280.

Varnham, K. (2006), Non-native species in UK Overseas Territories: a review. JNCC Report. Pp 372.

Verrill, A.E. (1902). *The Bermuda islands.* Transactions of the Connecticut Academy of Arts and Sciences*l:* New Haven: Connecticut, United States, 11:38.

Verrill, A.E. (1907). *The Bermuda Islands. Part IV. Geology and palaeontology, and Part V, An account of the coral reefs*. Transactions of the Connecticut Academy of Arts and Sciences: New Haven, Connecticut, United States, 12:45–348.

Vitt, L.J., and Cooper, W.E. (1986). Skink reproduction and sexual dimorphism: *Eumeces fasciatus* in the southeastern United States, with notes on *Eumeces inexpectatus*. *Journal of Herpetology,* 20:65–76.

Wallace, A.R. (1876). *The geographical distribution of animals: With a study of the relations of living and extinct faunas as elucidating the past changes of the earth's surface*. Macmillan & Co: London, United Kingdom.

Walther, G.R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J.C., Fromentin, J.M., Hoegh-Guldberg, O., and Bairlein, F. (2002). Ecological responses to recent climate change. *Nature*, 416:389–395.

Whitney, K.D., and Gabler, C.A. (2008). Rapid evolution in introduced species, 'invasive traits' and recipient communities: challenges for predicting invasive potential. *Diversity and Distributions*, 14:569–580.

Whittaker, R.J., and Fernandez-Palacios, J.M. (2007). *Island biogeography: ecology, evolution, and conservation*. Second Edition. Oxford University Press: Oxford, United Kingdom.

Wilkinson, H.C. (1973). *Bermuda from sail to steam: The history of the island from 1784 to 1901.* Volumes I and II. Oxford University Press: Oxford, United Kingdom. Pp 951.

Wingate, D.B. (1965). Terrestrial herpetofauna of Bermuda. *Herpetologica,* 21:202–218.

Wingate, R.S. (1998). *A comparison of demography and morphological variation in two insular populations of the Bermuda Rock Lizard (Eumeces longirostris).* MSc Thesis. University of Wales: Swansea, Wales.

Wingate, D.B. (2005). *Rare find of a clutch of skink eggs in the wild at Spittal Pond.* Critter Talk, Bermuda Aquarium, Museum and Zoo: Bermuda. Pp 28.

Wingate, D.B. (2011). The successful elimination of Cane toads, *Bufo marinus*, from an island with breeding habitat off Bermuda. *Biological Invasions,* 13:1487–1492.

Wright, G.R., and Eason, C.T. (1991). *Residue analysis of invertebrate samples for brodifacoum* (*Talon*)*.* Forest Research Institute Contract Report: Wellington, New Zealand.

Zachos, J., Pagani, M., Sloan, L., Thomas, E., and Billups, K. (2001). Trends, rhythms, and aberrations in global climate 65 Ma to present. *Science*, 292:686–693.

Zavaleta, E.S, Hobbs, R.J., and Mooney, H.A (2001). Viewing invasive species removal in a whole-ecosystem context. *Trends in Ecology and Evolution*, 16:454–459.

Zhang L., Hua, N., and Sun, S. (2008). Wildlife trade, consumption and conservation awareness in southwest China. *Biodiversity Conservation*, 17:1493–1516.

Zucker, N. (1994). A dual status-signalling system: a matter of redundancy or differing roles?. *Animal Behaviour,* 47:15–22.

# CHAPTER 2

# ESTIMATING POPULATION PARAMETERS FOR THE CRITICALLY ENDANGERED BERMUDA SKINK USING ROBUST DESIGN MARK-CAPTURE-RECAPTURE MODELLING

HELENA TURNER, RICHARD A. GRIFFITHS, MARK E. OUTERBRIDGE AND GERARDO GARCIA

Turner, H., Griffiths, R. A., Outerbridge, M. E., and Garcia, G. (2019). Estimating population parameters for the critically endangered Bermuda skink using robust design capture-mark-recapture modelling. *Oryx,* 1–8. DOI:10.1017/S0030605318001485.

Glass jars (4.5 litres) - the current trapping method for capturing Bermuda skinks. (*Photo Credit: Turner, H. 2016*).

## 2.1 ABSTRACT

Estimating reliable population parameters for highly secretive or critically endangered animals presents numerous challenges. We report on the status of the two largest remaining populations of critically endangered Bermuda skinks (*Plestiodon longirostris*), using a robust design capture-mark-recapture (CMR) analysis of PIT-tagged skinks on two islands over three years. The models provide precise estimates of abundance, capture and survival probabilities and temporary emigration. We estimated an abundance of *N* = 547 (± SE 63.5) skinks on Southampton Island and *N* = 277 (± SE 28.4) skinks on Castle Island. The populations do not appear to be stable and fluctuated at both sites over the three-year period. Despite these two seemingly healthy populations, the Bermuda skink remains threatened due to increasing anthropogenic activities, invasive species, and habitat loss.

## 2.2 INTRODUCTION

Many endangered species are cryptic, elusive and challenging to survey reliably. The development of efficient survey methods to detect them is vitally important for conservation management. A wide variety of methods have been used in monitoring studies, including capture-mark-recapture (CMR) techniques to estimate abundance and survival (Krebs, 1999; Besbeas et al., 2002), which are both critical determinants of population viability (White and Burnham, 1999). Conventionally, marking is used to uniquely identify individuals in successive samples. For lizards, photographic identification using natural markings (Sacchi et al., 2010), tagging using passive integrated transponder (PIT) tags (Germano and Williams, 1993), and individual recognition from DNA sampling (Moore et al., 2009) have increased the utility and application of the CMR approach.

The Bermuda skink (*Plestiodon longirostris*) is the country's only extant endemic reptile (Edgar et al., 2010). Bermuda skink populations have undergone significant declines in the past 50 years due to increased anthropogenic disturbances, habitat loss and degradation plus the introduction of invasive flora and fauna (Davenport et al., 1997; Glasspool and Outerbridge, 2004). The continued threats are likely to have a major impact on the remaining populations. These are already fragmented and isolated, and the lizards are now only found within a few nature reserves and offshore islands with ca. 2,300 - 3,500 individuals thought to be left on Bermuda (Edgar et al., 2010). However, as it has been over ten years since any CMR surveys were undertaken (Glasspool and Outerbridge, 2004) the current state of the population remains unclear.

We used a robust design model to monitor trends in abundance, survival, capture probabilities and temporary emigration, of the two largest skink populations on Bermuda. The robust design model of Pollock (1982) is an extension of the Cormack-Jolly-Seber (CJS) model and has become increasingly popular as it combines the advantages of both the live recapture model and the closed capture models. The model and its assumptions are described in detail by Cormack (1968), Kendall et al. (1995), Otis et al. (1978) and Seber (1982, 1986).

### *2.2.1 Study area*

This study was undertaken at two sites located within Castle Harbour, Bermuda (Fig. 2.1). Southampton Island (0.8 ha, 32.342 °N, -64.667 °W) and Castle Island (1.4 ha, 32.340 °N, -64.672 °W). These two sites were chosen because they are thought to represent the healthiest known skink populations (Hammond, 2000; Glasspool and Outerbridge, 2004), are both within protected nature reserves and are considered the most suitable and viable areas to conserve. The islands are separated by a water channel that is 330 m wide and are similar in abiotic factors. Castle Island is closer to the main island (164 m), and therefore faces more threats from invasive species and anthropogenic activities which have been the main reasons for the skink's decline across Bermuda. Additional sites across Bermuda were surveyed but sample sizes were too small (< 50 individuals), to provide reliable estimates.

## 2.3 METHODOLOGY

### *2.3.1 Skink capture and marking*

Methods were adapted from Davenport et al. (1997) using a consistent survey protocol across the two locations. Sixty-five, 4.5 litre glass jars (approximately 240 mm x 160 mm) were used as traps. Traps were set on 10 x 10 m grid systems because skink home ranges are approximately 10 $m^2$ (Davenport et al., 1997). Traps were mounted at a 45°-90° angle, with rocks or vegetation placed around so the skinks could gain access to the trap. Flagging tape was used to label each trap and locations were recorded using a handheld GPS (Garmin Etrex 20, +/- 15 m). Palmetto fronds and small towels were used to shade the traps. The pitfall traps were baited every day with 10 ml of canned sardines (BUMBLEBEE® sardines in oil) which were placed in a small sealed tea strainer to prevent consumption, which could affect recapture rates. Additionally, 5 ml of cod liver oil was smeared around the rim of the trap to prevent escape and to deter ants (Davenport et al., 1997). The traps took around two hours to pre-bait and were then checked hourly between 1100 – 1600 hrs. Traps were closed by removing the bait and turning the jar upside down to prevent accidental captures. Surveys were rescheduled if the temperature dropped below 21°C or heavy rainfall or winds (> 15 – 20 knots) were forecast as the lizards become less active (M. Outerbridge 2015, pers. comm., 11 March) under such conditions and landing on the islands in bad weather becomes challenging.

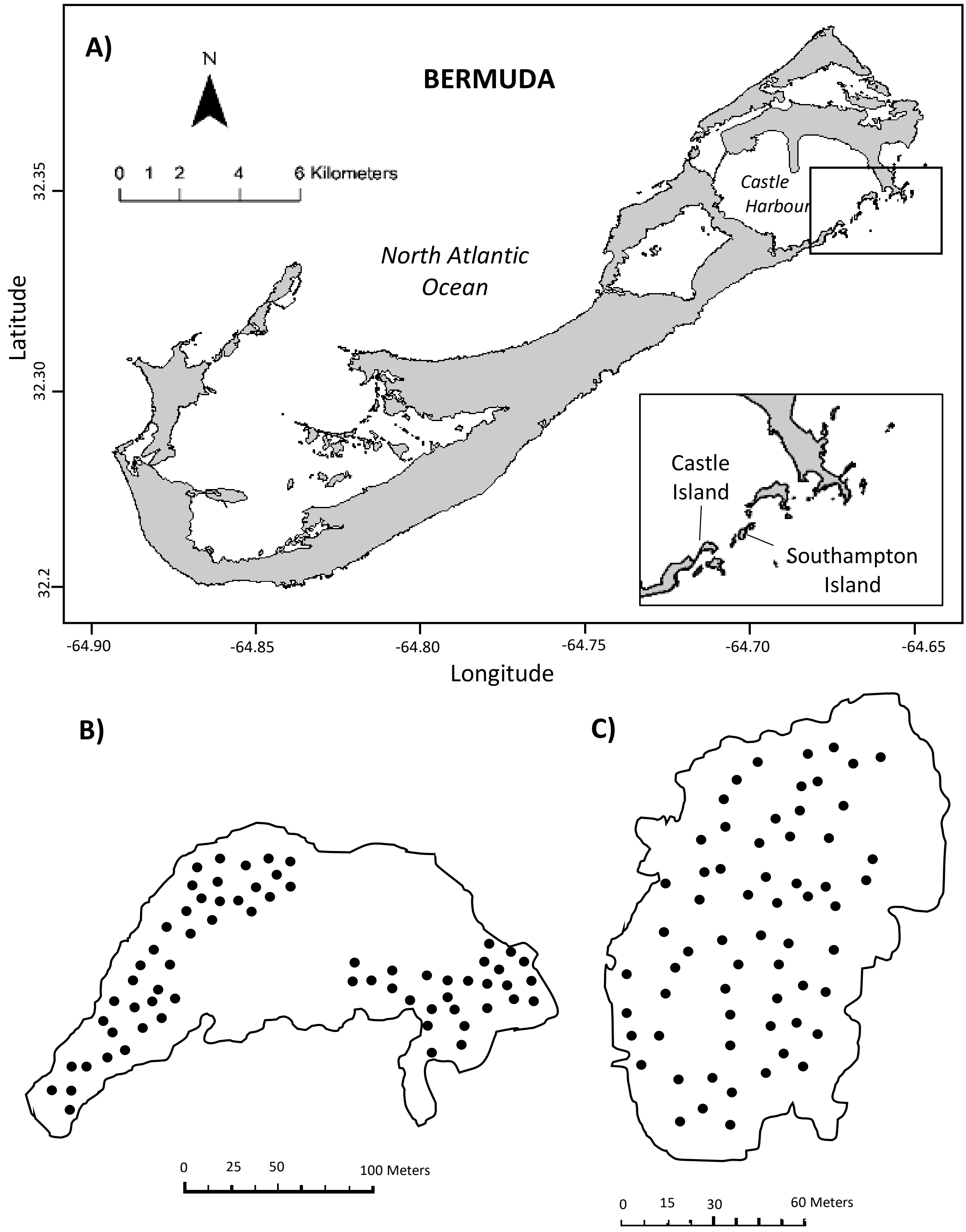


**FIGURE 2.1** A) Map of Bermuda with aerial view of trap locations B) On Castle Island and C) On Southampton Island, within Castle Harbour. (Image created in ArcMap 10.5).

On both islands data were collected over a five-day period in each month from May – July in 2015, 2016 and 2017, resulting in a total of 15 sampling occasions. Each sampling occasion consisted of five trap checks per day, with a total trapping time of 75 hours per year per island. At each capture, animals were weighed and measured (snout-vent length, SVL), unmarked animals were tagged, and tags of recaptured animals were recorded. All individuals (excluding juveniles: SVL < 64 mm, body mass < 11.5 g) in good health were tagged with a passive integrated transponder (PIT) tag. Any unmarked animals collected from the traps were implanted with tags (8 x 1.4 mm, 30 mg ID162B FDX-B Trovan®) operating at a frequency of 134.2 kHz. The tags were inserted subcutaneously in either the left or right lateral side of the body, using a syringe implanter (IM-200) with a 1.25" 14-gauge sterile disposable hypodermic needle. Prior to tagging, the injection site was wiped with F10® antiseptic solution. Immediately after injection, a drop of Loctite® cyanoacrylate glue was applied over the injection site to prevent tag loss and speed wound healing (Germano and Williams, 1993; Gibbons and Andrews, 2004). Once implanted, the tags were checked using a PIT tag reader (Universal (LID-560) scanner-Trovan®) which showed the individual's unique identification code. Even though the application of the internal tags may induce temporary stress (Langkilde and Shine, 2006), if inserted properly the tags are not known to cause any significant negative impacts (Ferner, 1979). Once processed, individuals were immediately released at the site of capture. (*See Supplementary Material 3 for a more detailed description of the marking technique*).

### *2.3.2 Statistical analyses*

Capture histories were compiled as time series of zeros and ones, where one stands for a capture success and zero for a non-capture using a standard 'X-matrix format' (Otis et al., 1978; Nichols, 1992). Rows represented the capture histories of each captured individual and columns represented the number of capture occasions. The CMR modelling was based on a robust design model (Nichols, 1992). This assumes the population to be closed to demographic parameters between secondary sampling occasions or days (i.e. no births, deaths, emigration, or immigration during each five day sampling period), but is assumed to be open to demographic changes between primary sampling occasions or years (over three years: 2015, 2016, 2017) where the population can experience these changes (Fig. 2.2). The resulting encounter history consisted of 15 capture occasions with unequal time spacing i.e. the five days were not always consecutive days nor was sampling undertaken on the same date each year due to weather or logistical challenges.

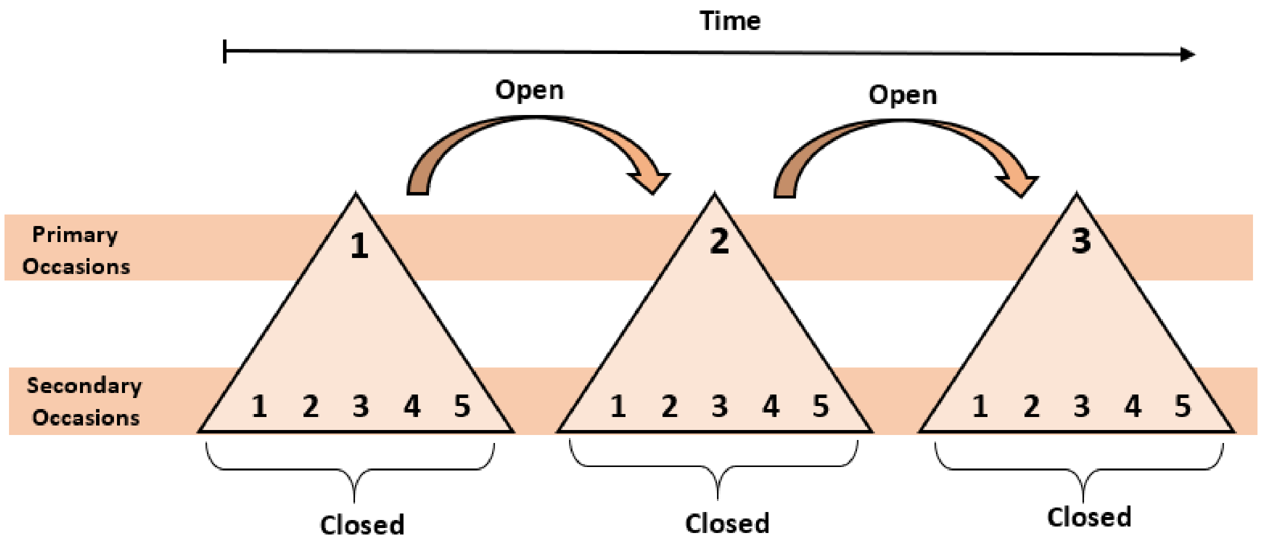


**FIGURE 2.2** Robust design example, with three primary trapping sessions each consisting of five secondary occasions.

### *2.3.3 Model selection*

To monitor population trends between the two skink populations the robust design model provided estimations based on the probability of survival ($\varphi$), probability of capture ($p$), temporary emigration ($\gamma$) and population size ($N$). This was undertaken using the packages 'marked' 'descr' and 'Rcapture' implemented in *R v. 3.4.3* (Rivest and Baillargeon, 2014; R Core Team, 2016). Estimates of the demographic parameters were derived using maximum likelihood estimates of the loglinear parameters with the R functions 'glm', and standard errors were calculated by linearisation (Rivest and Daigle, 2004; Baillargeon and Rivest, 2007).

Ten loglinear models were used to account for the ($t$) time and ($h$) heterogeneity effects of capture probabilities on the two islands (Table 2.1). These included the model with heterogeneity effects $M_h$ (the average probability of capture); the model with time effects $M_t$ (the capture probabilities for each capture occasion); the model with both time and heterogeneity effects $M_{th}$ (the average probabilities of capture for each occasion); and the null model with no time or heterogeneity effects $M_0$ (the capture probability at any capture occasion) (Rivest and Baillargeon, 2014). The $M_{th}$ and $M_h$ models were additionally fitted with four heterogeneity estimators: Chao (Chao, 1987), Poisson and Gamma (Rivest and Baillargeon, 2007) and Darroch (Darroch et al., 1993).

Temporary emigration was additionally tested for each island (between year one – year two and year two – year three), as the probability of capturing an individual may vary between capture occasions. The models were fitted and compared based on the Akaike information criterion (AIC) values.

## 2.4 RESULTS

Over three years 536 skinks were captured at both study sites, of these, 47 individuals were caught on Castle Island but were not tagged (43 too small and 4 escaped) and 85 individuals on Southampton Island were not tagged (66 too small and 19 escaped). As some individuals escaped during the capture event, detection was not recorded. Overall, 404 individuals were PIT tagged and used in further analyses. More than half of the marked skinks were recaptured on Southampton Island (52.6%; 133 of 253) and (58.3%; 88 of 151) on Castle Island over the three years (Appendix 2A).

The $\mathrm{M_{th}}$ model with Chao's estimator for each period has the lowest AIC and hence the best fit to the data for both islands (Table 2.1). According to the model, estimates of abundance on Southampton Island were $N = 547 \pm 63.5$ (SE) and estimates on Castle Island were $N = 277 \pm 28.4$ (SE). Over the three years, estimates appeared to fluctuate between years on both islands (Fig. 2.3). When comparing mean abundance between Southampton Island (mean = 253, SD = 105) and Castle Island (mean = 164.4, SD = 86.9) estimates followed a normal distribution ($P < 0.05$).

Estimates of annual capture probability derived using the model $\mathrm{M_{th}}$ were found to be slightly higher on Southampton Island (mean, $p = 0.50 \pm 0.04$ (SE)) compared to Castle Island (mean, $p = 0.42 \pm 0.06$ (SE)). Capture probabilities were lowest on Southampton Island in year one ($p = 0.44 \pm 0.05$) and highest in year two ($p = 0.57 \pm 0.09$). Similarly, on Castle Island, capture probabilities were lowest in year one ($p = 0.34 \pm 0.08$) and highest in year two ($p = 0.54 \pm 0.08$). The estimates were very precise (small standard errors) due to relatively high capture probabilities (Fig. 2.4).

**Table 2.1.**

Model selection criteria fit to Bermuda skink CMR data for Southampton Island and Castle Island, Bermuda. All models include abundance estimates and standard errors (SE).

| Location | Model | Estimate | SE | AIC | ΔAIC | df | Deviance |
|---|---|---|---|---|---|---|---|
| Southampton Island | $M_{th\ Chao}$ | 547.2 | 63.5 | 571.53 | 0.00 | 32740 | 332.39 |
| | $M_{th\ Poisson}$ | 595.5 | 80.4 | 574.31 | 2.78 | 32743 | 336.17 |
| | $M_{th\ Gamma}$ | 724.1 | 91.6 | 574.44 | 2.91 | 32745 | 340.30 |
| | $M_{th\ Darroch}$ | 719.0 | 86.3 | 576.44 | 4.91 | 32745 | 337.30 |
| | $M_t$ | 482.2 | 43.5 | 589.20 | 17.67 | 32746 | 357.06 |
| | $M_{h\ Darroch}$ | 718.7 | 86.2 | 607.07 | 35.54 | 32757 | 396.93 |
| | $M_{h\ Poisson}$ | 601.1 | 81.7 | 609.85 | 38.32 | 32755 | 395.72 |
| | $M_{h\ Gamma}$ | 722.5 | 91.3 | 609.86 | 38.33 | 32757 | 399.72 |
| | $M_{h\ Chao}$ | 555.2 | 65.0 | 612.34 | 40.81 | 32752 | 392.20 |
| | $M_0$ | 490.9 | 44.7 | 622.94 | 51.41 | 32758 | 414.80 |
| Castle Island | $M_{th\ Chao}$ | 294.8 | 31.5 | 416.95 | 0.00 | 32745 | 249.36 |
| | $M_t$ | 274.7 | 27.5 | 418.64 | 1.69 | 32747 | 251.66 |
| | $M_{th\ Gamma}$ | 196.6 | 41.8 | 420.45 | 3.50 | 32743 | 247.36 |
| | $M_{th\ Darroch}$ | 183.6 | 24.1 | 421.06 | 4.11 | 32743 | 247.79 |
| | $M_{th\ Poisson}$ | 175.2 | 30.7 | 421.55 | 4.60 | 32743 | 248.27 |
| | $M_0$ | 292.7 | 30.6 | 521.41 | 104.46 | 32759 | 380.12 |
| | $M_{h\ Gamma}$ | 193.8 | 37.1 | 523.69 | 106.74 | 32755 | 374.41 |
| | $M_{h\ Chao}$ | 294.8 | 31.5 | 523.73 | 106.78 | 32757 | 378.45 |
| | $M_{h\ Darroch}$ | 183.5 | 24.1 | 523.99 | 107.04 | 32755 | 374.71 |
| | $M_{t\ Poisson}$ | 176.9 | 32.5 | 524.27 | 107.32 | 32755 | 374.98 |

Annual survival of skinks on Southampton Island ($\varphi$ = 0.58 ± 0.21 (SE)) was higher than on Castle Island ($\varphi$ = 0.40 ± 0.09 (SE)). Survival was lower at both sites from 2015 – 16 but increased by 113.52% on Southampton Island and by 58.72% on Castle Island from 2016 – 17.

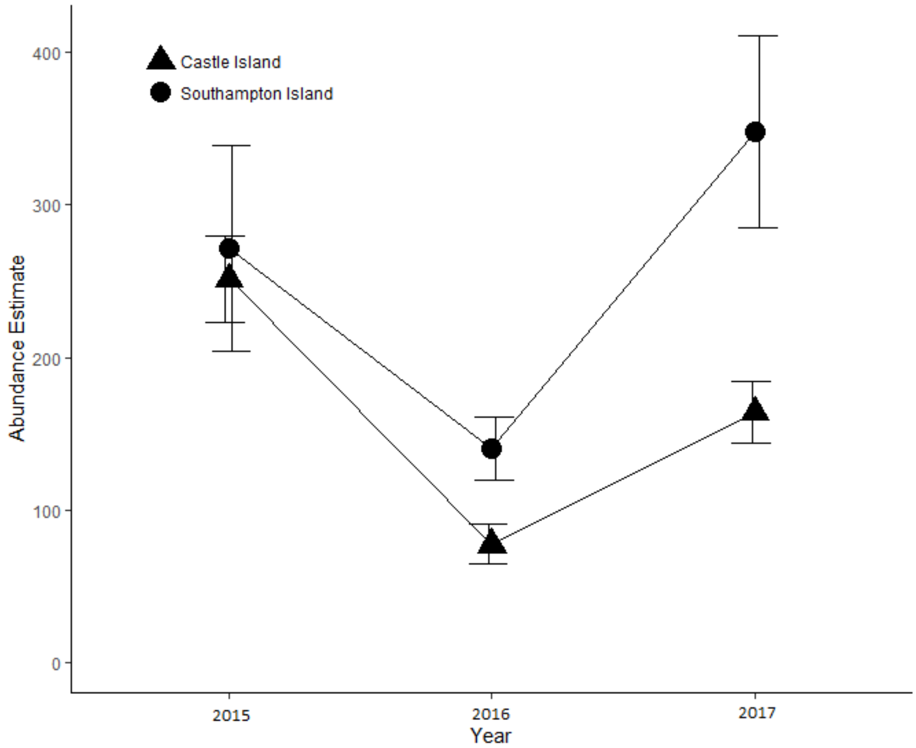


**FIGURE 2.3** Abundance estimation. Comparison of $\mathrm{M}_{\mathrm{th\ Chao}}$ robust design model of abundance estimates with standard error (vertical bars) of Bermuda skinks *Plestiodon longirostris* on Southampton Island and Castle Island, Bermuda between 2015 – 2017.

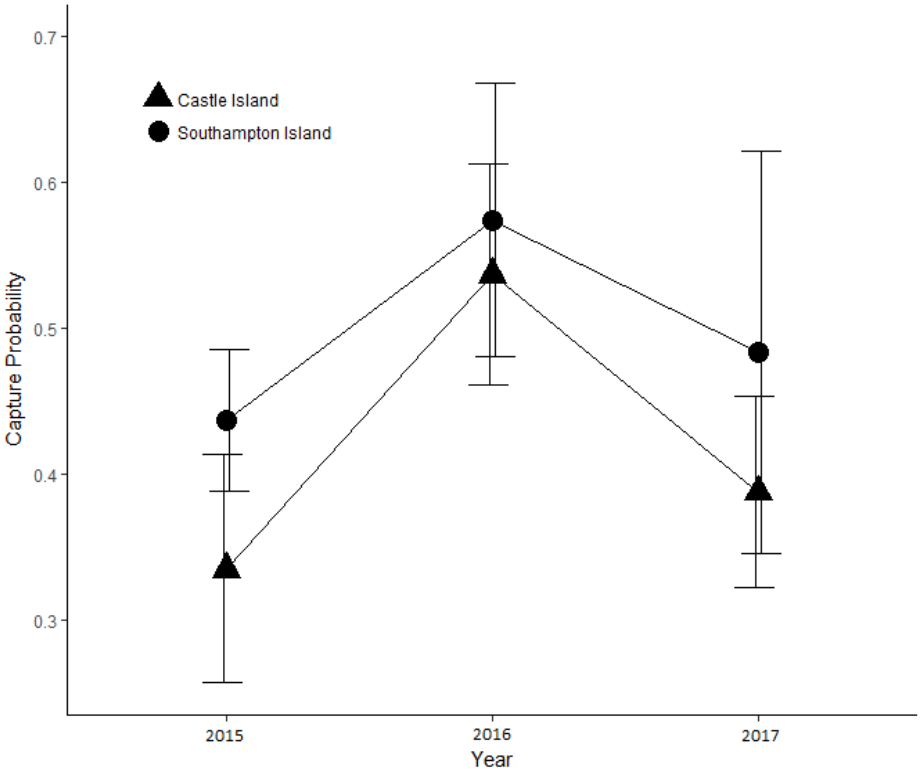


**FIGURE 2.4.** Capture probability. Comparison of $\mathrm{M}_{\mathrm{th\ Chao}}$ robust design model of capture probability with standard error (vertical bars) of Bermuda skinks *Plestiodon longirostris* on Southampton Island and Castle Island, Bermuda between 2015 – 2017.

When comparing models, the best fit models for both islands included temporary emigration between years, indicating that a small number of individuals were not available for capture within the sampling areas. Between 2015 and 2016, temporary emigration was highest on Southampton Island but was not apparent on Castle Island. Between 2016 and 2017 temporary emigration was higher on Southampton but also occurred on Castle Island (Table 2.2).

**TABLE 2.2.**
Estimated rates of Bermuda skink survival probability and temporary emigration with standard error (SE) between Southampton Island and Castle Island, Bermuda using a $\mathrm{M}_{\mathrm{th\ Chao}}$ robust design model between 2015 – 2017.

| Location | Sampling Period | Survival probability $\varphi$ ± SE | Temporary emigration $\gamma$ ± SE |
|---|---|---|---|
| Southampton Island | Year 1 (2015) → Year 2 (2016) | 0.372 ± 0.08 | 39.1 ± 32.2 |
| | Year 2 (2016) → Year 3 (2017) | 0.794 ± 0.19 | 36.7 ± 49.8 |
| Castle Island | Year 1 (2015) → Year 2 (2016) | 0.309 ± 0.06 | 0 ± 0.0 |
| | Year 2 (2016) → Year 3 (2017) | 0.490 ± 0.19 | 26 ± 11.9 |

## 2.5 DISCUSSION

### *2.5.1 Population trends over time*

The study provided precise estimates of abundance, capture and survival probabilities of a critically endangered lizard at two study sites. Our results demonstrate the advantages of using PIT tags, a reliable method for marking lizards long-term, as well as using a robust design model, as an effective approach to monitoring trends in skink populations.

Previous population estimates of Bermuda skinks were calculated using the simple Lincoln-Petersen method, which may be subject to bias (Seber, 1982). During previous surveys temporary marking methods (acrylic paint spots) were used, therefore recaptures may have been undercounted and the population over-estimated. Additionally, juveniles were not accounted for (due to the trapping method), and because studies were undertaken during the breeding season, brooding females were assumed unavailable for capture. Nevertheless, skink population sizes of 414 in 1997 (Davenport et al., 1997), and 582 in 2004 (Glasspool and

Outerbridge, 2004) were calculated for Southampton Island. On Castle Island 116 individuals were estimated in 2000 (Hammond, 2000). Population estimates were also calculated in 1998 at four other sites on Bermuda, where 44 individuals were estimated on Palm Island in Sandys Parish, 52 on Inner Pear Rock, 123 on Charles Island (both in St. George's Parish), and 124 at Spittal Pond a mainland nature reserve in Smith's Parish (Raine, 1998; Wingate, 1998). Although these populations showed reasonable proportions of juvenile and potential breeding individuals at the time, twenty years later none of them had large enough populations to have confidence in their long-term future (Davenport et al., 2001).

Estimating survival is important in demographic studies. For example, if skink recruitment is low then persistence, repeated breeding, and longevity may be a key factor in their recovery. At both sites there was a year of low survival between 2015 and 2016 which resulted in a population decrease, then a year of high survival between 2016 and 2017 which led to an increase in recruitment of the larger size classes. This was most likely the result of El Niño (a cyclic shift in atmospheric patterns) between 2015 and 2016. Although it resulted in fewer major impacts such as tropical storms and hurricanes affecting Bermuda, there was increased precipitation and stronger winds recorded during this time than in 2017 (Bermuda Weather Service, 2017). These unfavourable conditions would have led to fewer opportunities for foraging, basking, breeding, and incubating eggs as well as impacting seasonal food abundance.

Populations fluctuated widely on the two islands between years. However, fluctuations followed similar trends on the two islands. The exact causes of these synchronized fluctuations are unknown, but it is possible they result from variation in the availability of prey, impact of predators and climatic conditions. Alternatively, the fluctuations may be related to social interactions, resource availability, habitat quality, sampling variability or the populations even reaching environmental carrying capacity. Southampton Island continues to have the highest density of skinks in Bermuda, mostly because landing is prohibited and introduced predators and competitors (i.e rats *Rattus* sp., kiskadee flycatchers *Pitangus sulphuratus,* cats *Felis catus,* anolis lizards *Anolis* sp., and yellow-crowned night-herons *Nyctanassa violacea*), invasive plant species (i.e asparagus fern *Asparagus densiflorus* and casuarina trees *Casuarina equisetifolia*) and anthropogenic threats are largely absent.

However, these threats are present on Castle Island and may explain why abundance may take slightly longer to recover from dips (Fig. 3). Studying longer-term population dynamics may reveal whether these estimates are cyclic or stable, together with factors limiting population growth.

### *2.5.2 Meeting model assumptions*

Lizards may shift territories frequently during the breeding season (Ruby, 1978), or in response to fluctuations in food availability (Hews, 1993). Consequently, animals are more likely to be captured at some locations and times than others. This violates the standard assumptions of basic CMR models (Seber, 1982; Hammond, 1986). However, the interval between the primary capture trapping sessions (approximately one year) was sufficiently long to ensure that gains (births and immigration) and losses (deaths and emigration) would occur and the sample size collected was large enough to detect heterogeneity and the $M_{th}$ model to fit well.

Temporary emigration may occur when a proportion of the population remains unavailable for capture. For example, skinks often spend a considerable time within rock refugia where they are difficult to capture. During the breeding season many females may be sedentary due to nest guarding (Glasspool and Outerbridge, 2004) and are therefore unavailable for capture. If temporary emigration is not tested for it can lead to negatively biased population estimates and model assumptions may be violated (Hammond, 1986, 1990).

For the management of small, threatened populations, the potential ramifications of under-estimating (negative bias) or over-estimating (positive bias) abundance are clearly important. As the robust design uses two levels of sampling it allows for more parameters to be estimated and for finer control over the relative precision of each parameter (Kendall and Pollock, 1992; Kendall et al., 1995). The only major problem associated with the robust design is the large trapping effort required (Pollock, 1982), i.e. a minimum of five days at each site recommended (Otis et al., 1978; Nichols, 1992), and the consequent cost of sampling intensively. We therefore recommend this design for future CMR studies aimed at estimating reptile demographic parameters, particularly for those that need to detect population declines before they become critical.

Overall, we found the ten different models produced quite different abundance estimates, which ranged between 175.2 – 294.8 individuals on Castle Island and 418.2 – 724.1 individuals on Southampton Island (both excluding juveniles). However, the abundance estimates based on $\mathrm{M_{th}}$ are reliable and using Chao's estimator provided lower bound conservative estimates that produced better fits than other estimators such as Darroch, Poisson and Gamma that can be highly variable (Rivest and Daigle, 2004), especially in small scale CMR studies (Chao, 1987). To estimate abundance with little bias, capture probability must be relatively high (Otis et al., 1978; Burnham and Overton, 1979). Skinks had a moderate probability of capture on both Southampton and Castle Island (between 0.33 – 0.54 and 0.44 – 0.57, respectively), so the trapping method is adequate to describe the dynamics of these populations. In general, the $\mathrm{M_{th}}$ estimator works well if most individuals are captured many times and when the population size is estimated to be >100 individuals (Otis et al., 1978).

In 2016, population sizes were low at both sites. However, at this time capture probability was at its highest. Although a considerable number of skinks were caught during this time (243 individuals in total), 53% were recaptured individuals (compared to 12% in 2015 and 36% in 2017) which explains lower abundance estimates.

### *2.5.3 Conclusion*

Although two relatively large populations of skinks have been identified, the population fluctuations suggest that they remain vulnerable and may not be viable long-term without management, such as predator control (Towns, 1991; Newman, 1994; Towns and Ferreira, 2001; Reardon et al., 2012), habitat restoration (Towns, 1994; Webb and Shine, 2000) and the creation of suitable burrows (Souter et al., 2004). Such interventions may be especially needed during these periods of fluctuation or when population growth is slow. We recommend using robust design models for evaluating population parameters when samples are taken over multiple days and years, as these can provide timely insights into population trends and the mechanisms driving them, which has important implications for future conservation and research efforts.

**Acknowledgements** We thank the Government of Bermuda for providing all the necessary permissions and are very grateful for their continued support that made this work possible. We are also grateful to everyone who provided valuable input and assistance in the field, and to R. McCrea and E. Matechou for statistical assistance. This research was funded by J. Summers Shaw, Chester Zoo, the States of Jersey, the British Herpetological Society and the Bermuda Zoological Society.

**Author contributions** H.T. –analysis and writing of core text. H.T., R.G. –experimental and statistical design. H.T., R.G., M.O., G.G –field work. R.G., GG., M.O. –text review and input.

**Conflicts of interest** None.

**Ethical standards** This research was approved by Chester Zoo and the University of Kent Research and Ethics Committee and was conducted under permits issued by the Government of Bermuda’s Department of Environment and Natural Resources. Handling and tagging of skinks were undertaken in accordance with the conditions of the licence.

## 2.6 LITERATURE CITED

Baillargeon, S., and Rivest, L.P. (2007). Rcapture: Loglinear models for capture-recapture in R. *Journal of Statistical Software*, 19(5).

Bermuda Weather Service. (2017). *Climate Data*. [Online]. http://www.weather.bm/climate.asp [Accessed 10 May 2017].

Besbeas. P., Freeman, S.N., Morgan, B.J.T., and Catchpole, E.A. (2002). Integrating mark-recapture recovery and census data to estimate animal abundance and demographic parameters. *Biometrics,* 58:540–547.

Burnham, K.P., and Overton, W.S. (1979). Robust estimation of population size when capture probabilities vary among animals. *Ecology*, 60:927–936.

Chao, A. (1987). Estimating the population size for capture-recapture data with unequal catchability. *Biometrics*, 45:427–438.

Cormack, R.M. (1968). The statistics of capture-recapture methods. *Oceanography and Marine Biology Annual Review,* 6:455–506.

Darroch, S.E., Fienberg, G., Glonek, B., and Junker, B. (1993). A three-sample multiple capture-recapture approach to the census population estimation with heterogeneous catchability. *Journal of the American Statistical Association*, 88: 1137–1148.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). A study of populations of the Bermudian rock lizard (skink), *Eumeces longirostris*, cope (1861), on the islands of Nonsuch and Southampton, Bermuda. Government of Bermuda, Department of Agriculture and Fisheries, Bermuda.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (2001). Threats to the critically endangered endemic Bermudian skink *Eumeces longirostris*. *Oryx,* 35:332–339.

Edgar, P., Kitson, L., Glasspool, A., and Sarkis, S. (2010) Recovery plan for the Bermuda skink *Eumeces longirostris*. Government of Bermuda, Department of Conservation Services: Bermuda.

Ferner, J.W. (1979). A review of marking techniques for amphibians and reptiles. *SSAR Herpetological,* 9:41.

Germano, D., and Williams, D. (1993). Field evaluation of using passive integrated transponder (PIT) tags to permanently mark lizards. *Herpetological Review*, 24:54–56.

Gibbons, J.W., and Andrews, K.M. (2004). PIT tagging: simple technology at its best. *BioScience,* 54:447–454.

Glasspool, A., and Outerbridge, M. (2004). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island*, Castle Harbour. Bermuda Zoological Society: Bermuda.

Hammond, P.S. (1986) Estimating the size of naturally marked whale populations using capture-recapture techniques. *Reports of the international Whaling Commission* (special issue) 8:253–82.

Hammond, P.S. (1990). Heterogeneity in the Gulf of Maine? Estimating humpback whale population size when capture probabilities are not equal. *Reports of the international Whaling Commission* (special issue) 12:135–9.

Hammond, M.P. (2000). *An analysis of Bermuda Skink (Eucemes longirostris) populations on Castle Island, Bermuda*. Bermuda Aquarium, Museum and Zoo: Bermuda.

Hews, D.K. (1993). Food resources affect female distribution and male mating opportunities in the iguanian lizard *Uta palmeri*. *Animal Behaviour*, 46:279–291.

Kendall, W.L., and Pollock, K.H. (1992). The robust design in capture-recapture studies: a review and evaluation by Monte Carlo simulation. In: *Wildlife 2001: Populations.* McCullough DR., and Barrett RH (Eds.) Pp 31–43. Elsevier Science Publishers: London, United Kingdom.

Kendall, W.L., Pollock, K.H., and Brownie, C. (1995). A likelihood-based approach to capture-recapture estimation of demographic parameters under the robust design. *Biometrics,* 51:293–308.

Krebs, C.J. (1999). *Ecological Methodology.* Second Ed. Addison-Wesley Educational Publishers Inc: Menlo Park, California, United States.

Langkilde, T., and Shine. R. (2006). How much stress do researchers inflict on their study animals? A case study using a scincid lizard, *Eulamprus heatwolei*. *Journal of Experimental Biology* 209:1035–1043.

Moore, J., Charles, A., Daugherty, H., and Nelson., N.J. (2009) Large male advantage: phenotypic and genetic correlates of territoriality in tuatara. *Journal of Herpetology*, 43:570–578.

Newman, D.G. (1994). Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor’s skink, *Cyclodina macgregori*. *New Zealand Journal of Zoology,* 21:443–456.

Nichols, J.D. (1992) Capture-recapture models. *Bioscience*, 42:94–102.

Otis, D.L., Burnham, K.P., White, G.C., and Anderson, D.R. (1978). Statistical inference from capture data on closed animal populations. *Wildlife Monograph,* 62:1–135.

Pollock, K.H. (1982). A capture–recapture design robust to unequal probability of capture. *Journal of Wildlife Management*, 46:757–760.

R Core Team (2016). *R: A language and environment for statistical computing*. The R foundation for statistical computing, Vienna, Austria.

Raine, A. (1998) *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the critically endangered Bermuda Rock Lizard (Eumeces longirostris).* MSc thesis. University College: London, United Kingdom.

Reardon, J.T., Whitmore, N., Holmes, K.M., Judd, L.M., Hutcheon, A.D., Norbury, G., and Mackenzie, D.I. (2012). Predator control allows critically endangered lizards to recover on mainland New Zealand. *New Zealand Journal of Ecology*, 36:141–150.

Rivest, L.P., and Baillargeon, S. (2007). Applications and extensions of Chao's moment estimator for the size of a closed population. *Biometrics*, 63:999–1006.

Rivest, L.P., and Baillargeon, S. (2014). RCapture: Loglinear models for capture-recapture experiments. R package version 1.4-2. [Online] Available at: http://CRAN.R-project-org/package=Rcapture [Assessed 20 September 2017].

Rivest, L.P., and Daigle, G. (2004). Loglinear models for the robust design in mark–recapture experiments. *Biometrics*, 60:100–107.

Ruby, D.E. (1978). Seasonal changes in the territorial behavior of the iguanid lizard *Sceloporus jarrovi*. *Copeia*, 1978:430–438.

Sacchi, R., Scali, S., Pellitteri-Rosa, D., Pupin, F., Gentilli, A., Tettamanti, S., Cavigioli, L., Racina, L., Maiocchi, V., Galeotti, P., and Fasola, M. (2010) Photographic identification in reptiles: a matter of scales. *Amphibia-Reptilia*, 31:489–502.

Seber, G.A.F. (1982) *The estimation of animal abundance and related parameters*. Second Edition. Macmillian: New York, United States.

Seber, G.A.F. (1986) A review of estimating animal abundance. *Biometrics,* 42:267–292.

Souter, N.J., Bull, C.M., and Hutchinson, M.N. (2004). Adding burrows to enhance a population of the endangered pygmy blue tongue lizard, *Tiliqua adelaidensis*. *Biological Conservation*, 116: 403–408.

Towns, D.R. (1991). Response of lizard assemblages in the Mercury Islands, New Zealand, to removal of an introduced rodent: the kiore (*Rattus exulans*). *Journal of the Royal Society of New Zealand*, 21:119–136.

Towns, D.R. (1994). The role of ecological restoration in the conservation of Whitaker's skink (*Cyclodina whitakeri*), a rare New Zealand lizard (Lacertilia: Scincidae). *New Zealand Journal of Zoology*, 21:457–471.

Towns, D.R., and Ferreira, S.M. (2001). Conservation of New Zealand lizards (Lacertilia: Scincidae) by translocation of small populations. *Biological Conservation,* 98:211–222.

Webb, J.K., and Shine, R. (2000). Paving the way for habitat restoration: can artificial rocks restore degraded habitats of endangered reptiles? *Biological Conservation*, 92:93–99.

White, G.C., and Burnham, K.P. (1999) Program MARK: Survival estimation from populations of marked animals. *Bird Study*, 46:120–139.

Wingate, R.S. (1998) *A comparison of demography and morphological variation in two insular populations of the Bermuda rock lizard (Eumeces longirostris).* MSc thesis. University of Wales: Swansea, Wales.

## **APPENDIX 2A -** PIT Tagging Breakdown between 2015 and 2017.

| | | Year 1 | Year 2 | Year 3 |
|---|---|---|---|---|
| Castle Island | Individuals Caught | 112 | 22 | 17 |
| | Recaptured Individuals | 18 | 49 | 21 |
| | Not Tagged | 16 | 20 | 7 |
| | Escapees | 3 | 0 | 1 |
| Southampton Island | Individuals Caught | 90 | 57 | 106 |
| | Recaptured Individuals | 17 | 55 | 61 |
| | Not Tagged | 10 | 30 | 26 |
| | Escapees | 8 | 7 | 4 |

# CHAPTER 3

# DYNAMIC OCCUPANCY MODELLING TO DETERMINE THE STATUS OF A CRITICALLY ENDANGERED LIZARD

HELENA TURNER, RICHARD A. GRIFFITHS, MARK E. OUTERBRIDGE AND GERARDO GARCIA

Nonsuch Island within Castle Harbour, Bermuda. (*Photo Credit: Bermuda Aerial Media, 2017*).

## 3.1 ABSTRACT

Monitoring of cryptic or endangered species poses multiple challenges for population assessment and conservation. Failing to account for imperfect detection can lead to biased estimates and misleading inferences about population status. We used a dynamic occupancy model that explicitly accounted for occupancy, detectability, colonisation and local extinction as a tool for monitoring the critically endangered Bermuda skink (*Plestiodon longirostris*). Using a standard survey method across Bermuda, 40 sites were monitored between 2015 and 2017, and skinks were detected at 13 locations, two of which were new records. Ten observation level and site-specific covariates were combined and considered to influence occupancy and detectability.

Although Bermuda skinks were only observed at 20% of the sites, predicted occupancy was estimated to be about 25%. The probability of detection was $P$ = 0.45 and was positively influenced by the site being an island, the presence of seabirds, prickly pears and coastal habitat. However, skinks were unlikely to be detected on sites with cat and rat predators. Our study provides unbiased estimates of occupancy that can be used to compare the status of Bermuda skinks across different sites and as a basis for long term monitoring. We recommend that this information be used in setting priorities to ensure conservation of the species.

## 3.2 INTRODUCTION

Monitoring long-term population change in reptiles is an integral part of effective conservation research and management. Since censuses across whole geographical ranges are logistically difficult, in a study of occupancy, population monitoring focuses on assessing subsets of populations known as sites, and the basic parameter of interest is the proportion of those sites occupied by a species at a given time *t* (Kendall and White, 2009).

As occupancy models have a long history of use in ecological studies, this approach is frequently used to guide management decisions for endangered or threatened species (Linkie et al., 2007) especially as they are often less expensive to collect than full capture-mark-recapture surveys or entire demographic datasets, used to estimate abundance or population size (Kendall and White, 2009; MacKenzie et al., 2017). Data are often collected using a variety of direct and indirect sampling approaches to target a particular species (e.g. counts of faecal material, camera trapping, mist netting, vocalisations or environmental DNA (eDNA)) and define the proportion of an area where the species is present, without the need to mark individuals in the population. Compared to standard or 'static' occupancy models, dynamic occupancy models can be used to consider changes that may occur over time and space (Geissler and Fuller, 1986; MacKenzie et al., 2002; Royle and Nichols, 2003). Variation in detectability in time and space is one of the main sources of error in many monitoring studies (Yoccoz et al., 2001; Mackenzie et al., 2002). If studies fail to account for imperfect detection it can result in some sites appearing to be unoccupied that are actually occupied (i.e. 'false absences') which is common in elusive species where individuals may be missed or go undetected (Yoccoz et al., 2001; Thompson, 2004). Although occupancy alone is a useful state variable, extinction and colonisation probabilities estimated over time can also be modelled in relation to site characteristics. These can provide a greater understanding of the dynamic processes that drive changes in occupancy and allow stronger inferences on the patterns observed.

The Bermuda skink (*Plestiodon* (= *Eumeces*) *longirostris*) is a cryptic, critically endangered lizard (according to the IUCN; Conyers and Wingate, 1996) and the only endemic terrestrial vertebrate on Bermuda (Davenport et al., 2001). Once abundant across Bermuda, the population continues to decline as a result of habitat loss, anthropogenic activities and invasive species. Their decline was originally noticed in the early 1900s when it was reported that they were rarely seen on the mainland, but prevalent on offshore islands particularly around the coastal cliffs (Verrill, 1902). However, concern for their survival only began nearly a century later, when a campaign was undertaken through the Bermuda Zoological Society with more than 33,000 questionnaire surveys distributed island wide. Over 158 people reported skink sightings, some of which helped identify additional sites across the mainland but very few of these sightings were recent, with some dating back several years and were unlikely to still exist. Monitoring commenced shortly after, with a focus on the sites with recent sightings in an attempt to further understand the various population sizes and demographic characteristics (Raine, 1998; Wingate, 1998; Davenport et al., 2001; Glasspool and Outerbridge, 2004). Due to difficulties in monitoring Bermuda skinks, an accurate island-wide population status is lacking, although, given that it was roughly estimated that 2,300 – 3,500 skinks remained in Bermuda, it was suggested that 5,000 or more animals might be present if new sites were found (Edgar et al., 2010). These estimates were based on previous population studies undertaken between 1997 and 2004, in which Southampton Island was estimated to have between 600 and 800 individuals (Davenport et al., 1997; Glasspool and Outerbridge, 2004) and Nonsuch island less than 100 individuals (Davenport et al., 1997), and the results were extrapolated to all other sites. For example, apart from Spittal pond at least 16 other mainland sub-populations were believed to exist on the mainland so 1,000 – 1,800 adults were estimated. Similarly, there are six other islands within Castle harbour (Palm Island, Charles Island, Rushy Island, Horn Rock, Inner and Outer Pear Islands) in which another 600 – 800 skinks were estimated (Edgar et al., 2010). The estimates are a rough guide and likely overestimate the true population, so determining present day distribution and population status is urgently needed to inform conservation of the species.

This study used three years of presence-absence surveys across Bermuda to predict occupancy, colonisation, local extinction and detectability. In establishing estimates using a dynamic occupancy model we hope to inform future management actions and define areas of conservation importance throughout Bermuda.

## 3.3 METHODOLOGY

### *3.3.1 Sampling and field methods*

Forty locations were sampled across Bermuda (Fig. 3.1) using detection – non-detections, to survey for skinks between 2015 and 2017. Although Bermuda skinks are active all year round, surveys were undertaken between April – July when they are thought to be most active (Edgar et al., 2010). As it was not feasible to survey the entire population of Bermuda skinks, sites were selected based on historic records, the emphasis being on sites which had not been surveyed for more than ten years (Glasspool and Outerbridge, 2004). All locations were either isolated islets or islands or situated at least 100 m apart to ensure independence among sites.

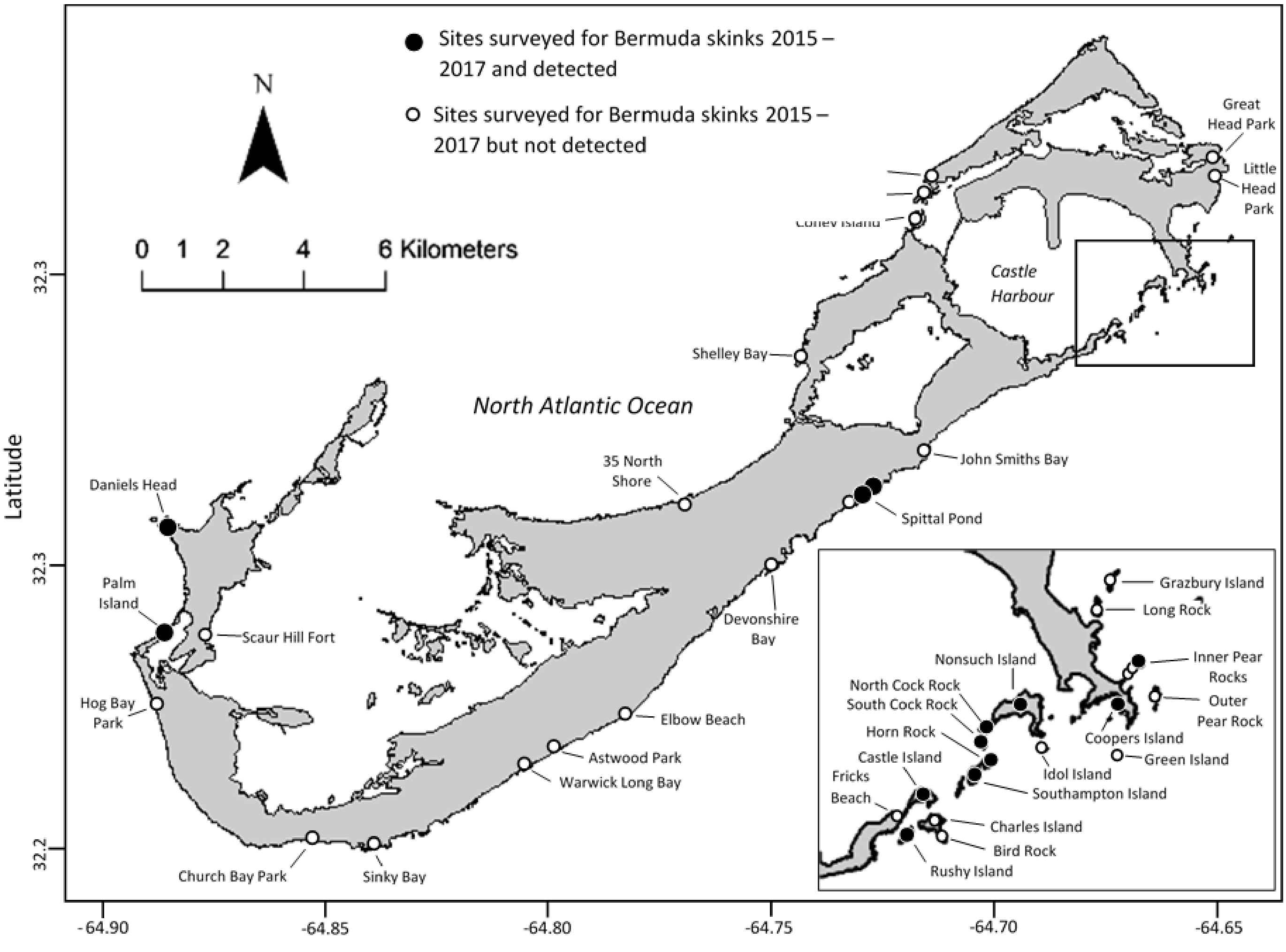


**FIGURE 3.1**. Detection and non-detection of Bermuda skinks across 40 localities on Bermuda 2015 – 2017.

(Image created in ArcMap 10.5).

The methodology followed that of Davenport et al. (1997) whereby skinks were initially captured using baited pit-fall traps. As the Bermuda skinks have small home ranges of around 10 $m^2$ (Davenport et al., 1997), traps were placed 5 - 20 m apart. Between 10 and 72 traps were set at each site, and more traps were set at larger sites to increase the chances of capturing skinks. Average trap density was 31 traps per site (0.008 traps per $m^2$; see Appendix 3A). The traps were deployed between 1000 hrs and 1100 hrs and opened between 1100 hrs and 1600 hrs (the skink's peak activity times; Davenport et al., 1997) with hourly checks to prevent trapped skinks from overheating, resulting in five rounds of checks per day. Each site was surveyed between two and fifteen times (depending on the weather and site access), for which skinks were either detected or not detected. To increase chances of detectability, surveys were not conducted during rain or high winds (> 40 kph), as suggested by Rhodes et al. (2006).

### *3.3.2 Statistical analyses*

All statistical analyses were undertaken using R v. 3.3.2 statistical software (R Core Team, 2016) and the function 'colext' in the package 'unmarked' (Fiske and Chandler, 2011). Occupancy modelling methods were applied using detection and non-detection data and a multi-season model to assess occupancy, colonisation, extinction, detectability and probable distribution of *P. longirostris* across Bermuda. The parameters are defined as: $\psi$ = probability of a site being occupied, $p$ = probability of being detected given presence, $\gamma$ = probability of colonisation and $\varepsilon$ = probability of local extinction. Within each year, model parameters $\psi$, $p$, $\gamma$, $\varepsilon$, must be constant, but changes in occupancy are modelled over time, from one year to the next and can be a function of covariates (i.e. site-specific habitat features). For each survey, model $p$, can vary among surveys with observation level covariates such as observers, survey equipment, environmental conditions, etc.

There are five critical assumptions that must be met during multi-season sampling for occupancy models (MacKenzie et al., 2005):

1) Another species is never wrongly identified as the target species.
2) Detection histories at each location are independent. Skinks must not be able to move between sites (i.e. the same individuals must not be detected at multiple sites).

3) There is no unmodelled heterogeneity in occupancy. Probability of occupancy is either constant across sites or differences are modelled using covariates.
4) There is no unmodelled heterogeneity in detection. Detectability is either constant across sites and surveys or differences are modelled using covariates.
5) Occupancy state is assumed ‘closed’. Occupancy status at each site does not change during the survey year, meaning the site is closed to changes in occupancy so skinks must be present at occupied sites for the duration of the survey year. This means no colonisation and extinction between secondary periods and no unmodelled heterogeneity in colonisation or extinction between primary periods.

If the assumptions are violated, estimates of occupancy, colonisation, extinction and detectability can be biased and inferences about factors that influence these parameters may be flawed (Gu and Swihart, 2004; MacKenzie et al., 2017).

### *3.3.3 Model covariates*

Occupancy probabilities may depend on covariates, which are incorporated into the models. Five site-specific covariates identified through previous studies (Davenport et al., 1997; Raine, 1998; Wingate, 1998; Glasspool and Outerbridge, 2004) and discussions with experienced local ecologists were included in the models (Table 3.1). These did not change with repeated visits and potentially explain site-to-site variability in occupancy, colonisation and extinction. Site-specific covariates included the site type (i.e. mainland or island), because anthropogenic disturbances on the mainland (i.e. coastal developments, coastal and beach activities and litter, which can be lethal) are known to threaten the skinks (Davenport et al., 1997, 2001; Raine, 1998; Wingate, 1998). Habitat type was included as the most prevalent habitat type at each site e.g. dense forest or coastal rock and scrub). As both nesting seabird colonies and seasonal fruits are thought to provide the skinks with an abundance of food (Davenport et al., 2001; Maderios, 2005). The presence of nesting seabirds (i.e. white-tailed tropicbirds *Phaethon lepturus catsbyii* and Bermuda petrels *Pterodroma cahow*) and prickly pears (*Opuntia dillenii*), were also included as site-specific covariates. Five selected introduced species (black and brown rats *Rattus* sp., kiskadee flycatchers *Pitangus sulphuratus*, cats *Felis catus*, Jamaican anoles *Anolis grahami* and Yellow-crowned night herons *Nyctanassa violacea*) were used as observation level covariates as these are associated with predatory threats to the skinks (see *Chapter 1* for background on introduced species).

**TABLE 3.1.**

Five site specific and five observation level predictor covariates influencing Bermuda skink site use during trapping occasions including descriptions, values and data classification type.

| | Covariate | Description | Value | Type |
|---|---|---|---|---|
| Site specific | Site Type (ST) | Mainland or Island site | 0 Mainland or 1 Island | Dichotomous |
| | Habitat Type (HT) | Site is mostly dense/forest habitat or coastal rock/scrub habitat | 0 Forest or 1 Coastal | Dichotomous |
| | Prickly Pears (PP) | Prickly pears present at site | 0 Absent or 1 Present | Dichotomous |
| | Seabird Nest (SN) | Nesting seabirds present at site | 0 Absent or 1 Present | Dichotomous |
| | No. of traps (NT) | Number of traps used | 10 - 72 | Discrete |
| Observation level | Presence of *Pitangus sulphuratus* (PK) | Kiskadees present at site | 0 Absent or 1 Present | Dichotomous |
| | Presence of *Felis catus* (PC) | Cats present at site | 0 Absent or 1 Present | Dichotomous |
| | Presence of *Nyctanassa violacea* (PH) | Yellow crowned night herons present at site | 0 Absent or 1 Present | Dichotomous |
| | Presence of *Anolis* sp. (PA) | Anoles present at site | 0 Absent or 1 Present | Dichotomous |
| | Presence of *Rattus* sp. (PR) | Rats present at site | 0 Absent or 1 Present | Dichotomous |

Although additional predators and competitors are present (*Anolis leachi*, *Anolis sagrei*, *Anolis extremus*, domestic chickens *Gallus domesticus*, and American crows *Corvus brachyrhynchos;* Wingate, 2011; Stroud et al., 2017) insufficient data were collected on their distribution to be included in this study. Visual cues were used to either confirm the presence or absence of each of the five observation level covariates during each trapping occasion. The number of traps used depended on the size of the site, therefore these were also included as covariates of detection.

### *3.3.4 Model selection and averaging*

In our first analysis (the null model), we assumed that all parameters (ｙ, ɣ, ε, p) were constant across sites and surveys and this was denoted by (.) to indicate that no covariates were being estimated. We used this model to provide a basic description for comparison with the unadjusted (i.e. naïve occupancy) proportion of sites where at least one skink was detected over the three-year survey period.

In a second analysis, we included covariates (MacKenzie et al., 2002, 2005) for each of the parameters that might affect occupancy, colonisation, extinction or detection probabilities using a maximum likelihood approach (MacKenzie et al., 2005). This allowed investigation into the relationships between the covariates, where estimates of occupancy may vary from year to year depending on covariates. All covariates were standardised (z-score) to ensure that each had equal predictive power and that the logit scale coefficients ($\hat{\beta}$) were not skewed by unevenly large ranges in the data (MacKenzie et al., 2005). Where logit scale coefficient ($\hat{\beta}$) estimates and associated standard errors (SE ($\hat{\beta}$)) are reported, a positive number suggests a positive relationship between the covariate and the model parameter it is a predictor of, and vice versa.

Logit scale coefficient ($\hat{\beta}$) estimates were back transformed using the ‘plogis’ function in R to give the model parameter estimates. Finally, using the package ‘AICcmodavg’, we assessed the fit of our models using a goodness-of-fit test (MacKenzie-Bailey test) based on bootstrapping (10,000 iterations) and Pearson’s $\chi^2$. The level of significance was set at P=0.05, with larger values indicating a lack of fit (MacKenzie and Bailey, 2004; Wright et al., 2016). To identify the most parsimonious and biologically plausible models for the observed data and assess which combination of covariates best explained the detection histories observed, we obtained and ranked Akaike’s Information Criterion (AIC) values for model comparison (Akaike, 1973; Burnham and Anderson, 2002). A set of candidate models was generated by selecting those that had a summed AIC weight of at least 0.95, indicating there was 95% confidence that these models best explained the data. If there were multiple top ranked models, a weighted model averaging technique was applied (Burnham and Anderson, 2002; Linkie et al., 2007) to estimate the occupancy, colonisation, local extinction and detection probabilities (with standard errors).

Finally, to optimise the survey design, the probability of detecting Bermuda skinks at least once if visiting a site *K* times was calculated from our best model to determine if the species is truly absent from a site, using the following mathematical expression (Pellet and Schmidt, 2005; Barata et al., 2017):

$$K = \log(1 - p^*) / \log(1 - p)$$

. Where $p$ is the detection probability; $p^*$ is the desired probability of detecting the species at an occupied site on at least one of the $K$ visits, and was set to 0.8, 0.9, 0.95 and 0.99 and log computes natural logarithms by default.

## 3.4 RESULTS

Out of the 40 sites surveyed, skinks were confirmed at 13 sites and sites were visited on average 5 times (range 2 – 15 visits). The distribution and frequency of captures are shown in Appendices 3A and 3B. The mean naïve occupancy (calculated by the number of sites presence was confirmed at) was 0.20 ± 0.06 (SE) over three years, 0.20 (8/40) in 2015, 0.30 (12/40) in 2016 and 0.10 (4/40) in 2017. The results showed that the naïve occupancy status did not remain constant across years, and estimates fluctuated across the sub-populations, with an increase in occupancy in year two. In comparison, the estimated probability of occupancy using the null model (AIC = 233.91) was 0.22 ± 0.08 (SE) across the three years. The difference between the naïve and the null model indicates that skinks may have been missed at 11% of sites, equating to three potential additional sites where skinks were present but remained undetected.

The top three models were selected as they had a cumulative weight of 0.96 indicating these models best explained the data (Table 3.2). Since the goodness-of-fit test gave a *P*-value lower than the level of significance (0.05), we accept the hypothesis that the models adequately fit the data (model 1: $\chi^2 = 205.4$, $P = 0.02$, model 2: $\chi^2 = 187.1$, $P = 0.03$, model 3: $\chi^2 = 140.5$, $P = 0.01$). The weighted model averaged estimates of occupancy ($\psi$), colonisation ($\gamma$), local extinction ($\varepsilon$) and detection ($p$) probabilities were taken as the final estimates. The estimate of occupancy (0.25 ± 0.06) did not vary considerably from the naïve occupancy estimate because the detection probability was relatively high (0.45 ± 0.06), therefore estimates are reasonably unbiased (MacKenzie et al., 2002; Table 3.3).

**TABLE 3.2.**

Relative differences in AIC values, delta AIC (Δ AIC), AIC model weights ($w_i$), the cumulative AIC model weights (cw) and the numbers of parameters in the models (*K*). Results are only presented for the top three ranked occupancy models with 95% confidence. Covariates include *PR*, presence of rats, *PC*, presence of cats, *HT*, Habitat type, *ST*, Site type, *NT*, No. of traps, *PP,* presence of prickly pears, *SB*, presence of nesting seabirds.

| Model No. | Occupancy Model Notation | AIC | Δ AIC | $w_i$ | cw | *K* |
|---|---|---|---|---|---|---|
| 1. | $\psi$ (*PP* + *ST* + *HT*), γ (*PC* + *SB*), $\varepsilon$ (*HT* + *PP*), $p$ (*SB* + *PR* + *PP* + CC + *HT*) | 168.12 | 0.00 | 0.71 | 0.71 | 12 |
| 2. | $\psi$ (*SB* + *PP* + *ST* + *HT*), γ (*PC* + *ST* + *SB*), $\varepsilon$ (*HT* + *PP*), $p$ (*PR* + *PP* + *HT*) | 171.33 | 3.21 | 0.14 | 0.85 | 16 |
| 3. | $\psi$ (.), γ (.), $\varepsilon$ (.), $p$ (*SB* + *PR* + *PP* + *PC* + *HT* + *ST)* | 173.69 | 5.57 | 0.11 | 0.96 | 10 |

Parameter estimates using the top model (AIC value = 168.12) suggest that there was a positive relationship between skink presence and coastal habitat type (2.44 ± 0.61 ($\hat{\beta}$ ± SE)), the presence of prickly pears (1.48 ± 0.55) and island sites (1.48 ± 0.55). The presence of cats (-0.32 ± 0.65) negatively influenced colonisation, whereas the presence of seabirds (3.10 ± 0.39) positively influenced colonisation. Local extinction was less likely in the presence of coastal habitat type (-3.10 ± 0.39) and presence of prickly pears (-3.10 ± 0.39). Detection was positively influenced by the presence of seabirds (7.10 ± 0.21), coastal habitat type (5.34 ± 0.22) and prickly pears (3.54 ± 0.54), whereas skinks were unlikely to be detected with the presence of rats (-2.94 ± 0.45) and even less so with cats (-4.30 ± 0.25). As all three of the top models were influenced by the same covariates and in the same directions, this suggests that there is a large amount of support for their inclusion. However, we found no support for the influence of the presence of kiskadees, herons or anoles, or the number of traps used on state variables.

**TABLE 3.3.**

Transformed beta ($\hat{\beta}$) estimates with weighted model averages. All associated standard errors (SE) are included. The first, second and third set of coefficients are for model terms associated with the occupancy (y), colonisation ($\gamma$) and local extinction ($\varepsilon$) parameters, respectively. The fourth set of coefficients explained heterogeneity in detection probabilities ($p$) associated with different revisits.

| Occupancy Model Notation | y ± SE | $\gamma$ ± SE | $\varepsilon$ ± SE | $p$ ± SE |
|---|---|---|---|---|
| ψ (*PP + ST + HT*), γ (*PC + SB*), $\varepsilon$ (*HT + PP*), $p$ (*SB + PR + PP +* CC *+ HT*) | 0.251 ± 0.05 | 0.128 ± 0.08 | 0.401 ± 0.08 | 0.456 ± 0.05 |
| ψ (*SB + PP + ST + HT*), γ (*PC + ST + SB*), $\varepsilon$ (*HT + PP*), $p$ (*PR + PP + HT*) | 0.249 ± 0.06 | 0.350.± 0.09 | 0.294 ± 0.07 | 0.444 ± 0.06 |
| ψ (.), γ (.), $\varepsilon$ (.), $p$ (*SB + PR + PP + PC + HT + ST)* | 0.236 ± 0.08 | 0.132 ± 0.06 | 0.312 ± 0.07 | 0.431 ± 0.09 |
| *Weighted model averages* | 0.248 ± 0.06 | 0.203 ± 0.08 | 0.351 ± 0.08 | 0.448 ± 0.06 |

### *3.4.1 Minimum number of surveys*

To improve the sampling protocol design, the number of surveys required to detect the Bermuda skink at a given site were calculated. Assuming skinks are imperfectly detected (detection probability <1), a detection probability between 80 and 99% was chosen to give the best chance to be able to monitor such a cryptic lizard. We found that 3 (mean: 2.74) visits were needed for an 80% probability of detecting skink presence, 4 (mean: 3.92) visits for a 90% probability of detecting skink presence, 5 (5.10) visits for a 95% probability and 8 (mean: 7.85) visits for a 99% probability that the Bermuda skink will be detected at a site (Fig. 3.2).

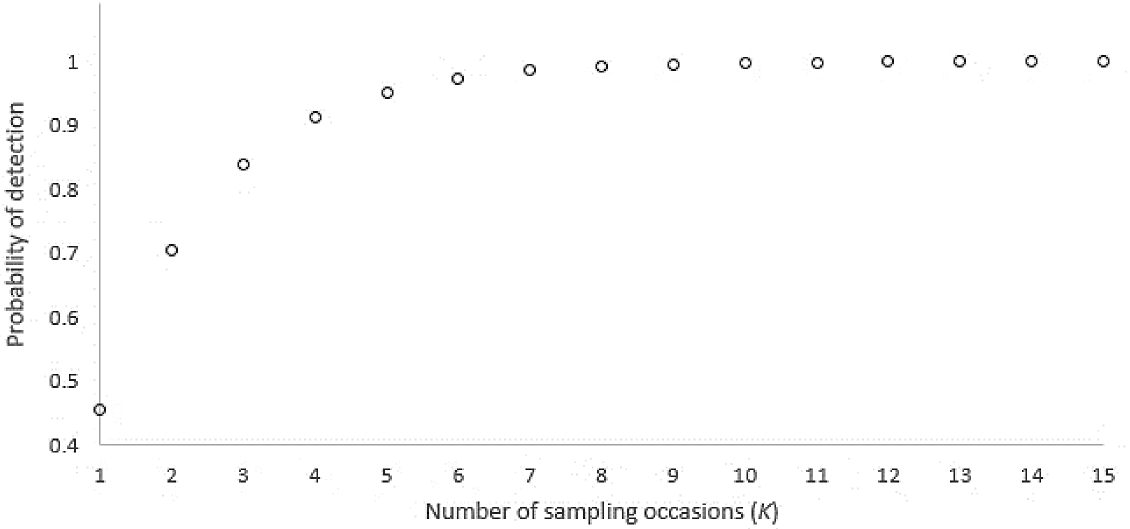


**FIGURE 3.2**. Probability of detecting Bermuda skinks at least once if visiting a site *K* times where model-averaged detection (*p)* = 0.448.

## 3.5 DISCUSSION

Conservationists increasingly rely on predictive models as a means for estimating patterns of species distribution. Monitoring programs are widely used to assess changes in wildlife populations, but they often implicitly assume constant detectability when documenting species occurrence (Pollock et al., 2002). This assumption is rarely met in practice because detectability varies across time and space. As a result, detectability of a species can be influenced by several physical, biological, or anthropogenic factors (e.g. weather, seasonality, topography, vegetation or even sampling methods). To evaluate some of these influences, we estimated site occupancy rates for *P. longirostris* using a multi-season model that explicitly takes detection, colonisation, extinction and occupancy probabilities into account, including a number of covariates assumed to be important in predicting skink occupancy, detection, colonisation and extinction.

### *3.5.1 Occupancy probability*

The estimated actual occupancy was higher than the naïve occupancy estimate, which indicates the importance of deriving robust estimates of parameters that take incomplete detectability into account (Pollock et al., 2002). Occupancy was related to the habitat type, the site type and the presence of prickly pears. This provides initial evidence that rocky coastal habitat and fortifications, particularly on offshore islands, continue to support and maintain skink populations (Glasspool and Outerbridge, 2004). Those sites with increased habitat degradation and habitat loss (especially on the mainland) were less likely to be occupied than islands (five mainland sites compared to nine island sites). This suggests that skinks require relatively undisturbed native habitats to thrive and they could therefore act as important biological indicators of the condition of coastal habitats in Bermuda.

The presence of scrub vegetation, such as prickly pears, may provide a seasonal supply of fruit; attract a plentiful supply of invertebrates; provide important shelter and refugia for skinks; and even play a key role in erosion control (Le Houérou, 1996). Raine (1998) also noted an association between skinks and areas dominated by coastal vegetation such as sea oxeye (*Borrichia arborescens*) and salt grass (*Spartina patens*) on Inner Pear, Charles Island and Spittal Pond.

Therefore, clearing areas of dense woodland (especially invasive vegetation such as Brazil pepper *Schinus terebinthifolia* and asparagus fern *Asparagus densiflorus*) in suitable coastal locations would be beneficial to skink survival. This action would simplify migration between population fragments and ultimately, increase the probability of population survival.

Although historical surveys conducted between 1998 and 2014 did not report on occupancy or detectability of the Bermuda skink, this study has shown that their range is declining because skinks were only detected at 12 of the 26 sites previously occupied in the past 20 years. Furthermore, only seven sites had more than one sampling occasion that detected a skink. As 67% of detections were located at the eastern end of Bermuda (i.e. within the Castle Harbour area), these populations remain extremely vulnerable to the continued threats from predation and habitat loss from hurricane damage. Those islands should therefore be provided with increased habitat management to control invasive species and reduce the harmful effects of litter (i.e. discarded empty bottles and cans), which can be lethal to the skinks (Jones, 2015).

### *3.5.2 Colonisation and extinction probabilities*

Seabirds were found to be important predictors of colonisation. The results confirm previous suggestions that skinks have a mutualistic relationship with Bermuda's nesting seabirds as the skinks will opportunistically forage in the seabird's nests, feeding on failed eggs, carrion such as dead chicks and dropped fish eaten by the skinks (Davenport et al., 1997). The installation of artificial nesting burrows has been an important component of the recovery of Bermuda's breeding white-tailed tropicbirds (*Phaethon lepturus catesbyi*) and Bermuda petrels (Madeiros, 2008), so the continuation of this process may help to sustain skinks and even encourage the colonisation of new skink sub-populations.

However, the probability of colonisation of skinks in the presence of cats will be highly unlikely, since cats have been observed predating skinks on many occasions (Garber, 1988). Domestic and feral cats have long been recognised by conservationists as one of the greatest threats to biodiversity (Woinarski et al., 2018) and are responsible for at least 14% of global bird, small mammal, and reptile extinctions and are the principal threat to almost 8% of critically endangered endemic species (Medina et al., 2011). In Australia, the average feral cat was reported to kill 225 reptiles per year (Woinarski et al., 2018). Therefore, with the growing number of domestic and especially feral cats in Bermuda (McGrath, 2014), a strategy describing all aspects of cat management on Bermuda, including the creation of a legislative and regulatory framework, is urgently needed.

Surprisingly, local extinction was not influenced by any predator covariates, but the lack of prickly pears and rocky coastal habitat were key factors. Therefore, changes in habitat are most likely to be important indicators in predicting local extinction. Average extinction probabilities tended to be higher than average colonisation probabilities. The variability in average colonisation and extinction probabilities suggest that populations are going through fluctuations in site occupancy. Population fluctuation is a common problem encountered in animal surveys. Monitoring occupancy can reveal changes in the status of a species over broad areas and may be appropriate for species that exhibit population fluctuations over short time periods.

### *3.5.3 Detectability probability*

We assessed the presence of multiple predator species alongside the detection of skinks at each site to determine which species pose threats to remnant skink populations. Although kiskadees, herons and anoles were thought to be a major problem for Bermuda skinks (Davenport et al., 1997), these were not covariates retained in the top models (anoles: AIC > 234.93, herons: AIC > 234.04 and kiskadees: AIC > 234.30). Birds are certainly capable of accessing all sites and Jamaican anoles are widespread across Bermuda (Stroud, et al., 2017), but they may not have always been observed at the sites because of the presence of humans. It is likely such predators are present more frequently at sites, but further information is needed to confirm this. However, the presence of rats and cats were found to negatively affect detection of the skinks. Continued monitoring of the prevalence of predators at each study site would determine the threat level they pose to the skinks and whether the management of introduced predators or competitors at a site would allow for effective recovery of skinks. Fortunately, introduced species can be eradicated, even from large islands. For example, coypu (*Myocastor coypus*) were eradicated from Great Britain (233,000 km$^2$; Gosling and Baker, 1989), rats (*Rattus norvegicus*) were eradicated from Langara Island, Canada (31 km$^2$; Taylor et al., 2000), and exotic herbivores are being removed from increasingly larger islands (Towns and Ballantine, 1993; Simberloff, 2001). Many of these projects benefited from the development of new poisoning and hunting techniques that have dramatically improved eradication techniques for goats (*Capra hircus*) (Campbell and Donlan, 2005), commensal rodents (*Rattus* sp. and *Mus musculus*) (Thomas and Taylor, 2002), rabbits (*Oryctolagus cuniculus*) (Chapuis et al., 2001), and pigs (*Sus scrofa*) (Cruz et al., 2005). In contrast, some invasive species such as cats, remain very difficult to eradicate from islands (Veitch, 2001). The largest island where cats have been successfully eradicated is Marion island, South Africa (290 km²), a project that took over 10 years (Bloomer and Bester, 1992). The second largest island where cat eradication has been successful is Little Barrier Island, New Zealand (28.1 km²); a project that took three years after previous failed attempts (Veitch, 2001). Now a rural village in New Zealand is even looking at banning domestic cats altogether to protect its native wildlife (Cooper, 2018). Unfortunately, there has not been many new developments in culling programs directed at managing these invasive populations. There is a strong reliance on broad scale use of chemical pesticides or other lethal methods of control. However, commonly used rodenticides such as brodifacoum are often toxic to the environment due to secondary and even tertiary poisoning (Mendenhall and Pank, 1980; Alterio, 1996; Eason et al., 1999). Therefore, a greater emphasis on ecologically based management would assist in reducing unintended effects of culling. Additionally, testing the success of different techniques to eradicate abundant invasives would greatly enhance current efforts to protect Bermuda’s native wildlife.

### *3.5.4 Improving parameter estimates*

Precise estimates of occupancy require a large number of sites, but for endangered species there may be insufficient sites for rigorous site replication. Due to poor weather and limited access, many of the sites in this study were visited only twice (resulting in an estimated 80% detectability). Consequently, lower precision with higher standard errors in estimates are likely. Therefore, increasing the number of sampling occasions to at least five should result in a 95% detection probability, to improve the precision of the estimated occupancy rate. However, there would be little gain in precision by undertaking more than five sampling occasions.

### *3.5.5 Future considerations*

Alternative methods to live-trapping skinks in the future could include the use of camera traps or drones with thermal imaging cameras for detecting these highly cryptic animals, particularly at rugged sites where the use of pitfall traps would not be feasible (Welbourne, 2013). Either motion-activated or devices on a timer can be deployed for consecutive days and have proved effective in detecting *Spenomorphus* skinks (Bennett and Clements, 2014). Both of these methods do have disadvantages. For example, drones at low altitude may disturb skinks, while drones at high altitude may be unable to detect the skinks or differentiate them from other non-native lizards, and they can be expensive especially for large-scale surveys. Nevertheless, the current traditional survey methods require significant human resources and bulky equipment, a boat to access sites, calm sea conditions to land personnel and equipment, and several days at each site to collect data. Additionally, the lizards need to be actively searching for food in order to enter traps. Camera traps and drones could circumvent some of these issues and incur minimal environmental disturbance (Henschel and Ray, 2003; Silveira et al., 2003). Equally, data from different methods can be combined in the same model to increase precision (Furnas and McGrann, 2018). Furthermore, they may provide the opportunity to collect additional information on habitat use (Henschel and Ray, 2003; Silveira at al. 2003), population structure, behaviour or diet (Silveira at el., 2003; Wegge et al., 2004), or even the presence of other species in the vicinity, which is not possible using the current method of pitfall trapping.

The use of dogs for detecting endangered or elusive species are also becoming a popular conservation method (Reed et al., 2011). Specially trained hunting dogs such as Jack Russell terriers are often trained to hunt or flush out rats or cats. Consequently, using sniffer dogs to target where species are present would greatly aid trap placement. Although dogs from elite hunting lines are often valued at around USD1000 – USD2000 each, they are often much easier to train and perform more effectively than non-hunting pet dogs (Wood et al., 2002).

Whichever method is chosen to detect species, future work should aim to achieve multiple annual presence/absence surveys at a variety of sites across Bermuda, in order to assess changes (e.g. trends) in site occupancy over time. Using models for estimating the dynamics of interacting species (e.g. competitors or predators) from replicated, presence-absence data would help to predict the current and potential distributions of invasive species which in turn could inform management decisions.

### *3.5.6 Conclusion*

Bermuda skinks exhibit high variation in occupancy and abundance between years (Turner et al., 2019). This seems to indicate high turnover of individuals, potentially indicative of a species with a high mortality rate or short life span. The Bermuda skinks had been thought to have a life span of around 21 years (Davenport et al., 1997). However, other closely related *Plestiodon* lizards have lifespans of 2–7 years in the wild (Clark et al., 2005). The patchy distribution of Bermuda skinks and the relatively high number of threats are additional constraints on dispersal and recruitment. As predators such as rats and cats clearly have a negative effect on the dynamics of the skink sub-populations, restoring native coastal habitat on offshore islands as well as the continuation of invasive species removal are priorities. Additionally, reviewing the success of survey methods regularly is beneficial to refining protocols and achieving the best trapping results.

## 3.6 LITERATURE CITED

Akaike, H. (1973). “Information theory as an extension of the maximum likelihood principle.” In: Petrov BN., and Csaki F (Eds.) *Second international symposium on information theory*. Akademiai Kiado: Budapest. Pp 267–281.

Alterio, N. (1996). Secondary poisoning of stoats (*Mustela erminea*), feral ferrets (*Mustela furo*), and feral house cats (*Felis catus*) by the anticoagulant poison, brodifacoum, New Zealand. *Journal of Zoology*, 23:331–338.

Barata, I.M., Griffiths, R.A., and Ridout, M.S. (2017). The power of monitoring: optimizing survey designs to detect occupancy changes in a rare amphibian population. *Scientific Reports*, 7:16491.

Bennett, D., and Clements, T. (2014). The use of passive infrared camera trapping systems in the study of frugivorous monitor lizards. *Biawak*, 8**:**19–30.

Bloomer, J.P., and Bester, M.N. (1992). Control of feral cats on sub-Antarctic Marion Island, Indian Ocean. *Biological Conservation*, 60:211-219.

Burnham, K.P., and Anderson, D.R. (2002). Model selection and multi model inference: A practical information-theoretic approach. Second Edition. Springer-Verlag: United States.

Campbell, K., and Donlan, C.J. (2005). Feral goat eradications on islands. *Conservation Biology*, 19:1362–1374.

Chapuis, J.L., Le Roux, V., Asseline, J., Lefevre, J., and Kerleau, F. (2001). Eradication of rabbits (*Oryctolagus cuniculus*) by poisoning on three islands of the subantarctic Kerguellen Archipelago. *Wildlife Research*, 28:323–331.

Clark, K., Hendricks, A., and Burge, D. (2005). Molecular identification and analysis of *Borrelia burgdorferi sensu lato* in lizards in the Southeastern United States. *Applied Environmental Microbiology*, 71:2616–2625.

Conyers, J., and Wingate, D. (1996). *Plestiodon longirostris. The IUCN red list of threatened species 1996.* [Online] Available at: http://dx.doi.org/10.2305/IUCN.UK.1996.RLTS.T8218A12900393.en [Assessed 23 August 2017].

Cooper, K. (2018). *Why a village in New Zealand is trying to ban all cats*. [Online]. Available at: https://www.bbc.co.uk/news/world-asia-45347136 [Assessed 13 April 2019].

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

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). *A study of populations of the Bermudian rock lizard (skink), Eumeces longirostris, cope (1861), on the islands of Nonsuch and Southampton, Bermuda*. Department of Agriculture and Fisheries: Bermuda.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (2001). Threats to the critically endangered endemic Bermudian skink *Eumeces longirostris*. *Oryx*, 35:332–339.

Eason, C.T., Milne, L., Potts, M., Morriss, G., Wright, G.R.G., and Sutherland, O.R.W. (1999). Secondary and tertiary poisoning risks associated with brodifacoum. *New Zealand Journal of Ecology,* 23:219–224.

Edgar, P., Kitson, L., Glasspool, A., and Sarkis, S. (2010). Bermuda skink recovery plan. Government of Bermuda, Department of Conservation Services: Bermuda.

Fiske, I.J., and Chandler, R.B. (2011). unmarked: An R package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software,* 43:10.

Furnas, B.J., and McGrann, M.C. (2018). Using occupancy modelling to monitor dates of peak vocal activity for passerines in California. *Condor*, 120:188–200.

Garber, S.D. (1988). Behaviour and ecology of the endangered endemic: Bermuda rock lizard (*Eumeces longirostris*). In: *Proceedings of the 12th International Symposium on Captive Propagation and Husbandry*. Pp 67–69.

Geissler, P.H., and Fuller, M.R. (1986). Estimation of the proportion of an area occupied by an animal species. In: *Proceedings of the American Statistical Association on Survey Research Methods*. Pp 533–537.

Glasspool, A., and Outerbridge, M. (2004). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island, Castle Harbour, Bermuda*. Bermuda Zoological Society: Bermuda.

Gosling, L.M., and Baker, S.J. (1989).The eradication of muskrats and coypus from Britain. *Biological Journal of the Linnean Society*, 38:39–51.

Gu, W., and Swihart, R.K. (2004). Absent or undetected? Effects of non-detection of species occurrence on wildlife–habitat models. *Biological Conservation,* 116:195–203.

Henschel, P., and Ray, J.C. (2003). *Leopards in African rainforests: survey and monitoring techniques*. Wildlife Conservation Society: United States.

Hudson, S. (1996). Natural toe loss in south-eastern Australian skinks: implications for marking lizards by toe clipping. *Journal of Herpetology*, 30:106–110.

Jones, S. (2015). *Litter a 'death trap' for skinks*. [Online]. Available at: http://www.royalgazette.com/article/20150702/NEWS07/150709939 [Assessed 9 March 2019].

Kendall, W.L., and White. G.C. (2009). A cautionary note on substituting spatial subunits for repeated temporal sampling in studies of site occupancy. *Journal of Applied Ecology,* 46:1182–1188.

Le Houérou, H.N. (1996). The role of cacti (*Opuntia* spp.) in erosion control, land reclamation, rehabilitation and agricultural development in the Mediterranean Basin. *Journal of Arid Environments*, 33:135–159.

Linkie, M., Dinatab, Y., Nugrohob A., and Haidirc, I.A. (2007). Estimating occupancy of a data deficient mammalian species living in tropical rainforests: Sun bears in the Kerinci Seblat region, Sumatra. *Biological Conservation*, 137:20 –27.

MacKenzie, D.I., Nichols, J.D., Lachman, G.B., Droege, S., Royle, J.A., and Langtimm, C.A. (2002). Estimating site occupancy rates when detection probabilities are less than one. *Ecology*, 83:2248–2255.

MacKenzie, D.I., and Bailey, L.L. (2004). Assessing the fit of site-occupancy models. *The Journal of Agricultural, Biological and Environmental Statistics,* 9:300–318.

MacKenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., and Hines, J.E. (2005). Occupancy estimation and modelling: Inferring patterns and dynamics of species occurrence. Elsevier Academic Press: United States

MacKenzie, D.I., Nichols, J.D., Royle, J.A., Pollock, K.H., Bailey, L.L., and Hines, J.E. (2017). Occupancy estimation and modelling: Inferring patterns and dynamics of species occurrence. Second Edition. Elsevier Academic Press: United States.

Madeiros, J. (2005). Recovery plan for the Bermuda petrel (cahow) *Pterodroma cahow*. Department of Conservation Services: Bermuda.

Madeiros, J. (2008). Breeding success survey of white-tailed tropicbird *Phaethon lepturus catsbyii* on the islands of Bermuda – 2006 to 2008 nesting seasons and comparison of breeding success between natural and artificial 'igloo' nest sites. Department of Conservation Services: Bermuda.

McGrath, L. (2014). *Feral cats a 'serious threat'*. [Online]. Available at: http://www.royalgazette.com/article/20140922/NEWS/140929966 [Assessed 9 March 2019]

Medina, F.M., Bonnaud, E., Vidal, E., Tershey, B.R., Zavaleta, E.S., Donlan, J., Keitt, B.S., Corre, M., Horwath, S.V., and Nogale, M.S. (2011). A global review of the impacts of invasive cats on island endangered vertebrates. *Global Change Biology*, 17:3503–3510.

Mendenhall, V.M., and Pank, L.F. (1980). Secondary poisoning of owls by anticoagulant rodenticides. *Wildlife Society Bulletin,* 8:311–315.

Pellet, J., and Schmidt, B.R. (2005). Monitoring distributions using call surveys: estimating site occupancy, detection probabilities and inferring absence. *Biological Conservation*, 123:27–35.

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

R Core Team. (2016). *R: A language and environment for statistical computing*. The R foundation for statistical computing: Vienna, Austria.

Raine, A. (1998). *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the critically endangered Bermuda Rock Lizard (Eumeces longirostris)*. MSc. Dissertation. University College London: United Kingdom.

Reed, S.E., Bidlack, A.L., Hurt, A. and Getz, W.M. (2011). Detection distance and environmental factors in conservation detection dog surveys. *The Journal of Wildlife Management*, 75:243–251.

Rhodes, J.R., Tyre, A.J., Jonzén, N., McAlpine, C.A., and Possingham, H.P. (2006). Optimizing presence–absence surveys for detecting population trends. *Journal of Wildlife Management*, 7:8–18.

Royle, J.A., and J.D. Nichols. (2003). Estimating abundance from repeated presence-absence data or point counts. *Ecology*, 84:777–790.

Silveira, L., Jacomo, A.T.A., and Diniz-Filho, J.A.F. (2003). Camera trap, line transect census and track surveys: a comparative evaluation. *Biological Conservation*, 114:351–355.

Simberloff, D. (2001). Eradication of island invasives: practical actions and results achieved. *Trends in Ecology and Evolution*, 16:273–274.

Stroud, J.T., Giery, S.T., and Outerbridge, M.E. (2017). Establishment of *Anolis sagrei* on Bermuda represents a novel ecological threat to critically endangered Bermuda skinks (*Plestiodon longirostris*). *Biological Invasions*, 19:1723–1731.

Thomas, W., and Taylor, R.H. (1993). "A history of ground-based rodent eradication techniques developed in New Zealand, 1959-1993." In: Veitch CR & Clout MN (Eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, United Kingdom. Pp. 301–310.

Thompson, W.L. (2004). "Introduction to sampling rare or elusive species." In: Thompson WL (Ed.) *Sampling rare or elusive species*. Island Press: United States. Pp 1–7.

Towns, D.R., and Ballantine, W.J. (1993). Conservation and restoration of New Zealand island ecosystems. *Trends in Ecology and Evolution*, 8:452–457.

Towns, D.R., and Broome, K.G. (2003). From small Maria to massive Campbell: Forty years of rat eradications from New Zealand islands. *New Zealand Journal of Zoology*, 30:377–398.

Turner, H., Griffiths, R. A., Outerbridge, M. E., and Garcia, G. (2019). Estimating population parameters for the critically endangered Bermuda skink using robust design capture-mark-recapture modelling. *Oryx,* 1–8.

Veitch, C.R. (2001). The eradication of feral cats (*Felis catus*) from Little Barrier Island, New Zealand. *New Zealand Journal of Zoology*, 28:1–12.

Verrill, A.E. (1902). *The Bermuda islands.* Transactions of the Connecticut Academy of Arts and Sciences*I:* New Haven: Connecticut, United States, 11:38.

Wegge, P., Pokheral, C.P., and Jnawali, S.R. (2004). Effects of trapping effort and trap shyness on estimates of tiger abundance from camera trap studies. *Animal Conservation*, 7:251–256.

Welbourne, D. (2013). A method for surveying diurnal terrestrial reptiles with passive infrared automatically triggered cameras. *Herpetological Review*, 44:247–250.

Wingate, R.S. (1998). *A comparison of demography and morphological variation in two insular populations of the Bermuda Rock Lizard* (*Eumeces longirostris*). BSc Dissertation. University of Wales: Wales.

Wingate, D.B. (2011). The successful elimination of Cane toads, *Bufo marinus*, from an island with breeding habitat off Bermuda. *Biological Invasions,* 13:1487–1492.

Woinarski, J.C.Z., Murphy, B.P., Palmer, R., Legge S.M., Dickman, C.R., Doherty, T.S., Edwards, G., Nankivell, A., Read, J.L., and Stokeld, D. (2018). How many reptiles are killed by cats in Australia? *Wildlife Research*, 45:247–266.

Wood, B., Tershy, B.R., Hermosillo, M.A., Donlan, C.J., Sanchez, J.A., Keitt, B.S., Croll, D.A., Howald, G.R., and Biavaschi, N. (2002). "Removing cats from islands in north-west Mexico." In: Veitch CR and Clout MN (Eds.). *Turning the tide: the eradication of invasive species*. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, United Kingdom. Pp. 374–380

Wright, W.J, Irvine, K.M., and Rodhouse, T.J. (2016). A goodness-of-fit test for occupancy models with correlated within-season revisits. *Ecology and Evolution*, 6:5404–5415.

Yoccoz, N.G., Nichols, J.D., and Boulinier, T. (2001). Monitoring of biological diversity in space and time. *Trends in Ecology and Evolution*, 16:446–453.

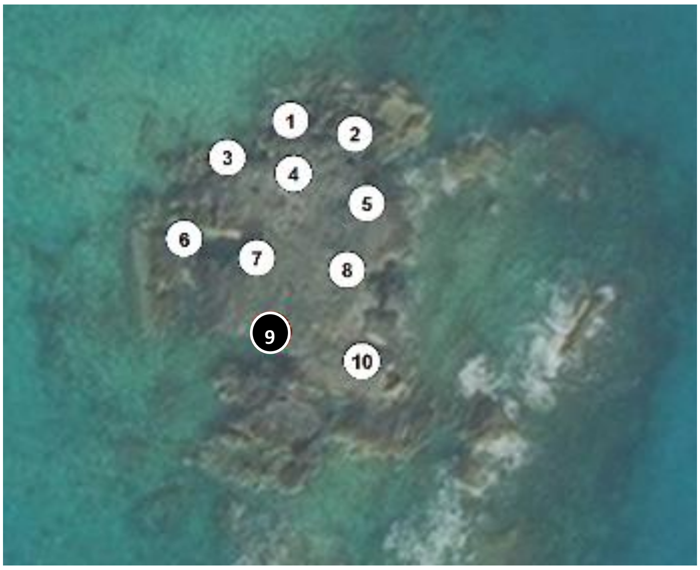
## **APPENDIX 3A** – Catch Per Unit Effort

Trapping summary at each site between 2015 and 2017: number of traps set during each survey, area surveyed (in square meters / acres), total number of skinks caught, number of surveyors, number of days and hours spent surveying, total trap hours, trap density (number of traps/land area) and catch per unit effort (CPUE) required to catch a skink. In comparison CPUE on Southampton Island in 1997 was 14.2 trap hours per skink (Davenport et al., 1997) and 2.95 trap hours per skink in 2004 (Glasspool and Outerbridge. 2004) and 8.9 trap hours per skink on Nonsuch Island (Davenport et al., 1997).

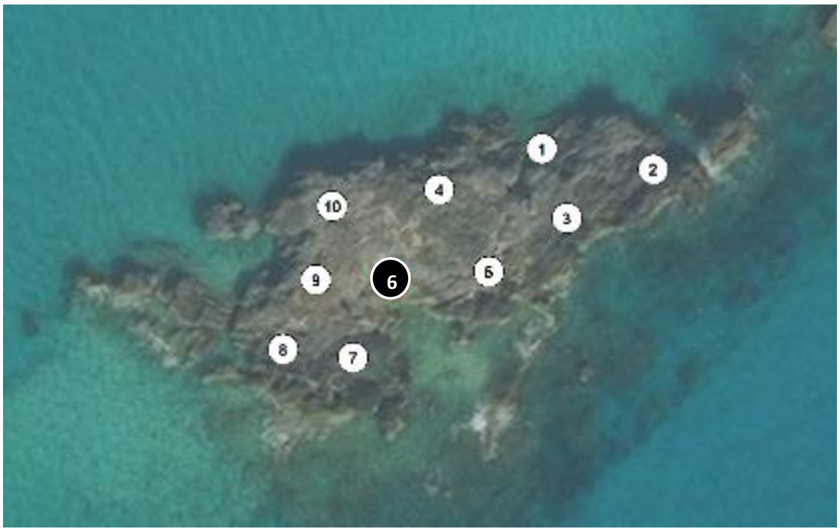
| | Site | No. of Traps | Area surveyed (m$^2$ / acres) | No. of skinks caught | No of surveyors | No of days (hours) surveyed | Trap hours | Trap Density (m$^2$ / acres) | CPUE |
|---|---|---|---|---|---|---|---|---|---|
| 1. | North Cock Rock | 10 | 1457 / 0.36 | 2 | 1 | 2 (10) | 100 | 0.00686 / 27.78 | 50 |
| 2. | South Cock Rock | 10 | 850 / 0.21 | 1 | 1 | 2 (10) | 100 | 0.01176 / 47.62 | 100 |
| 3. | Inner (Inner) Pear Rock | 4 | 405 / 0.10 | 0 | 3 | 2 (10) | 40 | 0.00988 / 40.00 | - |
| 4. | Ferry Point Island | 10 | 5301 / 1.31 | 0 | 3 | 2 (10) | 100 | 0.00188 /7.63 | - |
| 5. | Bird Rock | 15 | 1174 / 0.29 | 0 | 1 | 2 (10) | 150 | 0.01277 /34.48 | - |
| 6. | Idol Island | 15 | 728 / 0.18 | 0 | 1 | 2 (10) | 150 | 0.02060 / 83.33 | - |
| 7. | Scaur Hill Fort | 20 | 4856 / 1.20 | 0 | 1 | 2 (10) | 200 | 0.00411 /16.67 | - |
| 8. | East Elbow Beach | 20 | 5463 / 1.35 | 0 | 1 | 2 (10) | 200 | 0.00366 / 14.81 | - |
| 9. | Shelley Bay | 20 | 4452 / 1.10 | 0 | 1 | 2 (10) | 200 | 0.00449 /18.18 | - |
| 10. | Spittal Pond (Jeffries Hole) | 21 | 2954 / 0.73 | 37 | 4 | 15 (75) | 1575 | 0.0071 / 28.76 | 42.57 |
| 11. | Spittal Pond (Checkerboard) | 21 | 7325 / 1.81 | 0 | 4 | 15 (75) | 1575 | 0.00286 / 21.73 | - |
| 12. | Inner (Middle) Pear | 19 | 890 / 0.22 | 0 | 3 | 2 (10) | 190 | 0.0213 / 86.36 | - |
| 13. | Sinky Bay | 25 | 3440 / 0.85 | 0 | 3 | 6 (30) | 750 | 0.0073 / 29.41 | - |
| 14. | Long Rock | 25 | 5463 / 1.35 | 0 | 3 | 2 (10) | 250 | 0.00457 / 18.52 | - |
| 15. | Rushy Island | 25 | 3237 / 0.80 | 1 | 2 | 2 (10) | 250 | 0.00772 /31.25 | 250 |
| 16. | Coney Island | 25 | 8498 / 2.10 | 0 | 2 | 2 (10) | 250 | 0.00294 / 11.09 | - |
| 17. | Charles Island | 25 | 11979 / 2.96 | 0 | 2 | 6 (30) | 750 | 0.00209 /8.45 | - |
| 18. | Grazbury Island | 25 | 3399 / 0.84 | 0 | 3 | 2 (10) | 250 | 0.00736 / 29.76 | - |
| 19. | Inner (Outer) Pear | 25 | 1578 / 0.39 | 2 | 3 | 2 (10) | 250 | 0.0158 /64.10 | 125 |
| 20. | Outer Pear Rock | 25 | 1821 / 0.45 | 0 | 3 | 2 (10) | 250 | 0.01373/55.56 | - |
| 21. | Hog Bay Park | 27 | 6111 / 1.51 | 0 | 3 | 2 (10) | 270 | 0.01419 /17.88 | - |
| 22. | Astwood Park | 25 | 4249 / 1.05 | 0 | 3 | 2 (10) | 250 | 0.00588/23.81 | - |
| 23. | Horn Rock | 27 | 1902 / 0.47 | 2 | 3 | 6 (30) | 810 | 0.01419/57.45 | 405 |
| 24. | Church Bay | 30 | 4087 / 1.01 | 0 | 3 | 2 (10) | 300 | 0.00734/29.70 | - |
| 25. | John Smiths Bay | 30 | 4452 / 1.10 | 0 | 2 | 2 (10) | 300 | 0.00674/27.27 | - |
| 26. | Warwick Long Bay | 30 | 13152 / 3.25 | 0 | 3 | 2 (10) | 300 | 0.00228/9.23 | - |
| 27. | Green Island | 30 | 2226 / 0.55 | 0 | 3 | 2 (10) | 300 | 0.0135/54.55 | - |
| 28. | Ferry Point Park | 20 | 15378 / 3.80 | 0 | 3 | 2 (10) | 200 | 0.00130/ 5.26 | - |
| 29. | 35 North Shore | 30 | 2711 / 0.67 | 0 | 2 | 2 (10) | 300 | 0.0111/44.77 | - |
| 30. | Devonshire Bay Park | 30 | 9915 / 2.45 | 0 | 2 | 2 (10) | 300 | 0.00303/12.24 | - |
| 31. | Nonsuch Island | 72 | 57142 / 14.12 | 97 | 4 | 15 (75) | 2400 | 0.00126/5.10 | 24.74 |
| 32. | Spittal Pond (East End) | 33 | 9551 / 2.36 | 8 | 4 | 15 (75) | 2475 | 0.003455/13.98 | 309.38 |
| 33. | Daniels Head | 35 | 7487 / 1.85 | 2 | 2 | 2 (10) | 350 | 0.004675/18.92 | 175 |
| 34. | Great Head Park | 40 | 9186 / 2.27 | 0 | 3 | 2 (10) | 400 | 0.00435/17.62 | - |
| 35. | Little Head Park | 40 | 9389 / 2.32 | 0 | 3 | 2 (10) | 400 | 0.00426/17.24 | - |
| 36. | Coopers Island | 45 | 12950 / 3.20 | 2 | 3 | 6 (30) | 1350 | 0.00347/14.06 | 675 |
| 37. | Fricks Beach | 60 | 4006 / 0.99 | 0 | 3 | 2 (10) | 600 | 0.0150/60.61 | - |
| 38. | Palm Island | 65 | 2671 / 0.66 | 1 | 2 | 6 (30) | 1950 | 0.02436/98.48 | 1950 |
| 39. | Castle Island | 65 | 16309 / 4.03 | 327 | 4 | 15 (75) | 4875 | 0.00399/16.13 | 14.91 |
| 40. | Southampton Island | 65 | 8903 / 2.20 | 692 | 4 | 15 (75) | 4875 | 0.0073/29.55 | 7.04 |

## APPENDIX 3B – Aerial images of spatial placement and numbering of traps. Black circles indicate a skink was captured (Images edited in ArcMap 10.5).

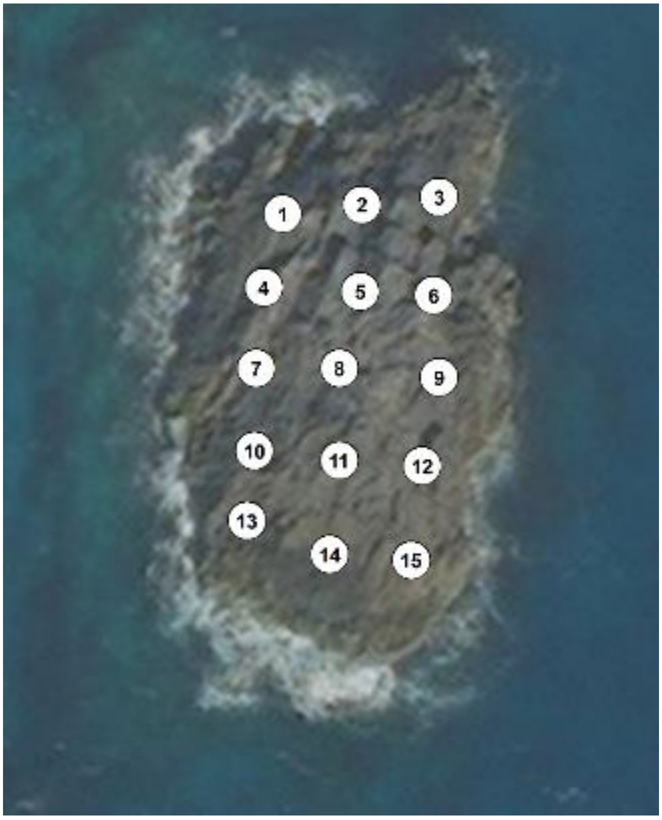
Aerial image of North Cock Rock showing spatial placement and numbering of traps (10 traps set).



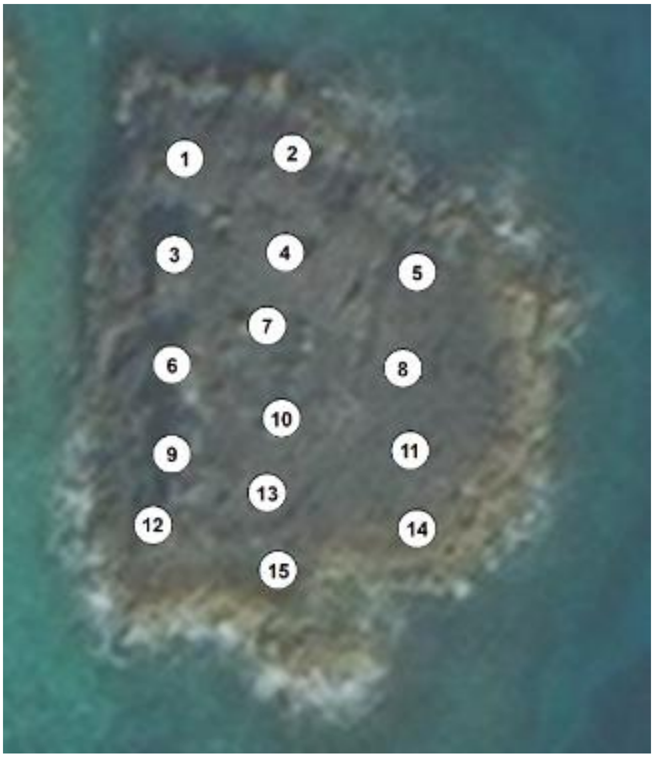
Aerial image of South Cock Rock showing spatial placement and numbering of traps (10 traps set).



Aerial image of Bird Rock showing spatial placement and numbering of traps (15 traps set).



Aerial image of Idol Island showing spatial placement and numbering of traps (15 traps set).



Aerial image of Fricks Beach showing spatial placement and numbering of traps (60 traps set).

Aerial image of Charles Island showing spatial placement and numbering of traps (25 traps set).

Aerial image of Coney Island showing spatial placement and numbering of traps (25 traps set).

Aerial image of Coopers Island showing spatial placement and numbering of traps (45 traps set).

Aerial image of Scaur Hill Fort Park showing spatial placement and numbering of traps (20 traps set).

Aerial image of Daniels Head showing spatial placement and numbering of traps (35 traps set).

Aerial image of Devonshire Bay showing spatial placement and numbering of traps (30 traps set).

Aerial image of Elbow Beach showing spatial placement and numbering of traps (20 traps set).

Aerial image of Ferry Point Park and Ferry Point Island showing spatial placement and numbering of traps (30 traps set).

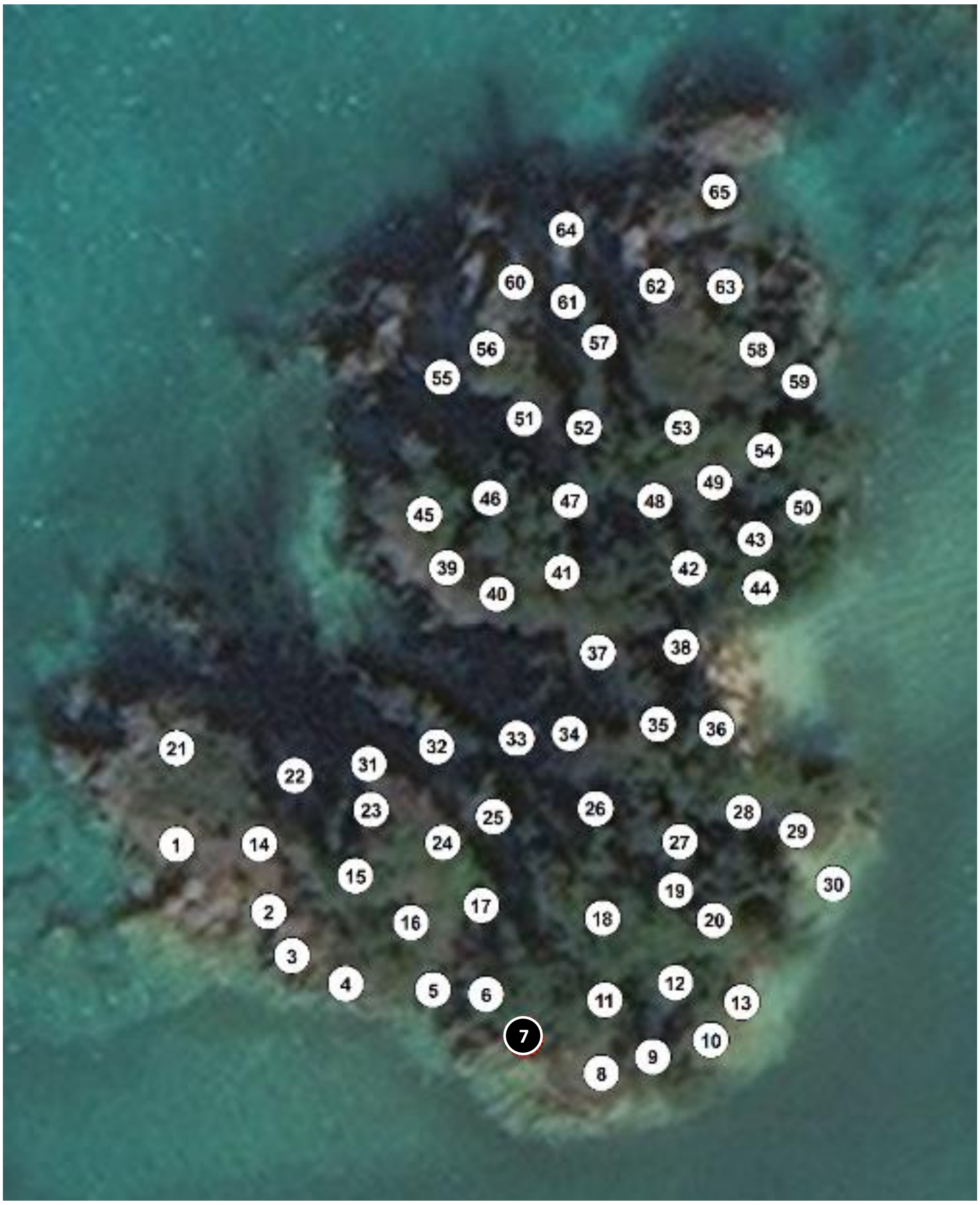
Aerial image of Shelley Bay showing spatial placement and numbering of traps (20 traps set).

Aerial image of Hog Bay Park showing spatial placement and numbering of traps (27 traps set).

Aerial image of Rushy Island showing spatial placement and numbering of traps (25 traps set).

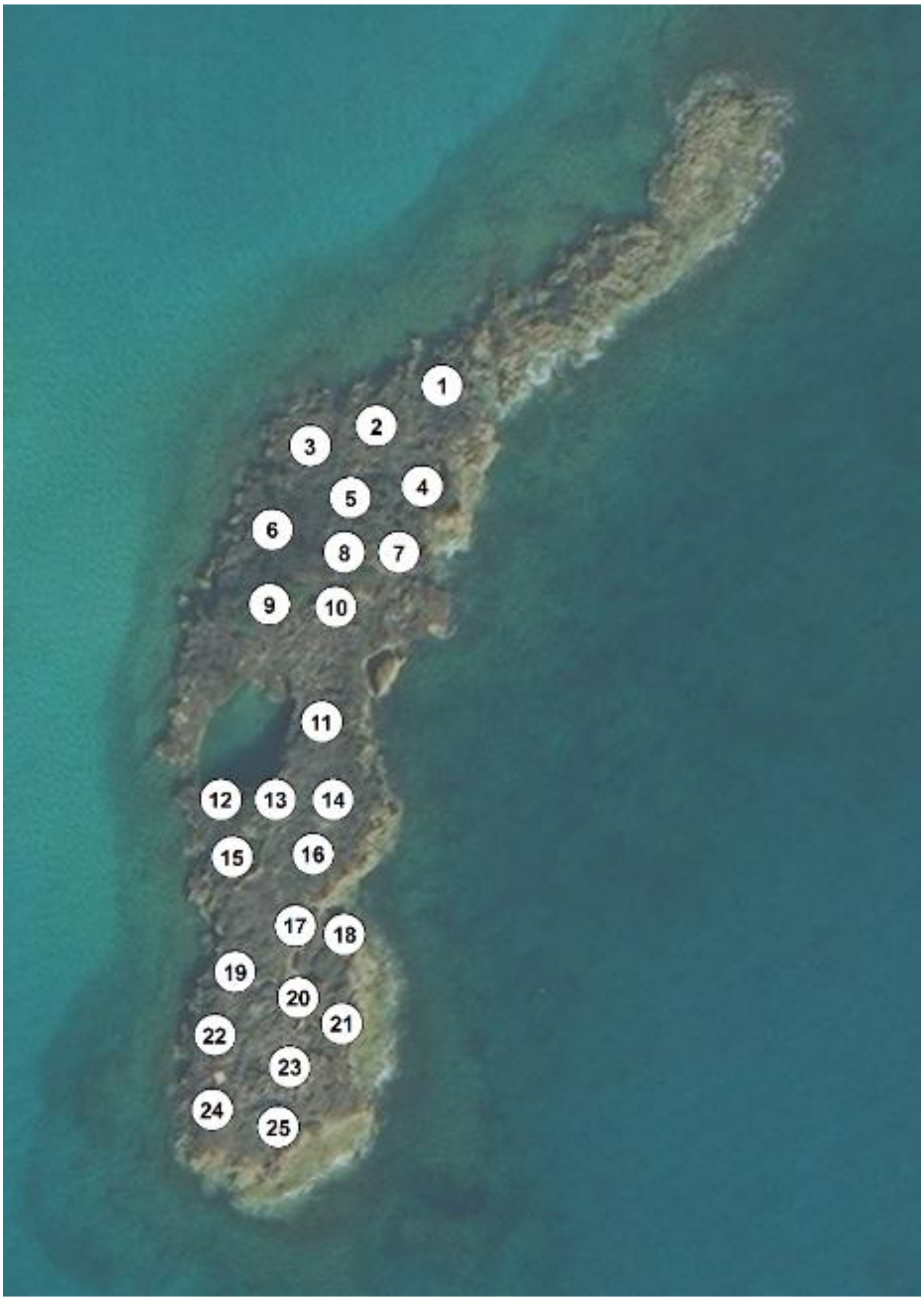
Aerial image of Sinky Bay showing spatial placement and numbering of traps (25 traps set).

Aerial image of Palm Island showing spatial placement and numbering of traps (65 traps set).



Aerial image of Astwood Park showing spatial placement and numbering of traps (25 traps set).

Aerial image of Long Rock showing spatial placement and numbering of traps (25 traps set).

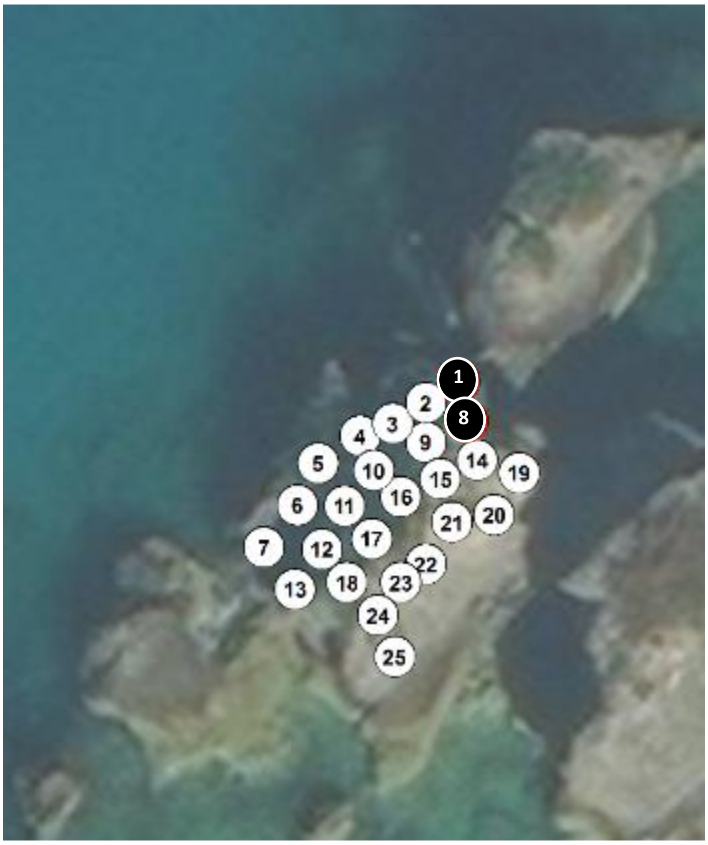


Aerial image of Grazbury's Island showing spatial placement and numbering of traps (25 traps set).

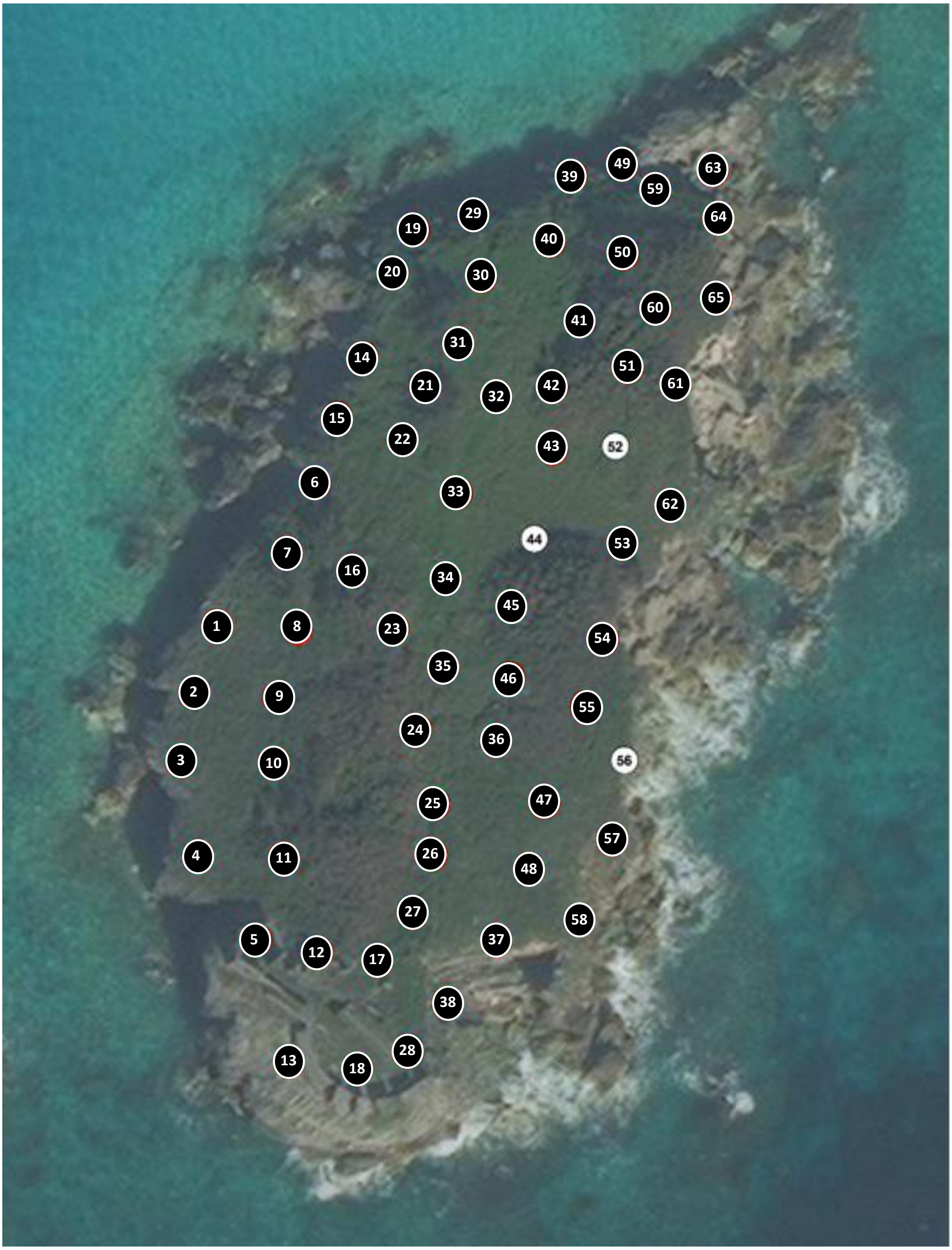
Aerial image of Little Head Park showing spatial placement and numbering of traps (40 traps set).

Aerial image of 35 North Shore showing spatial placement and numbering of traps (30 traps set).

Aerial image of Inner (Outer) Pear Rock showing spatial placement and numbering of traps (25 traps set).



Aerial image of Southampton Island showing spatial placement and numbering of traps (65 traps set).

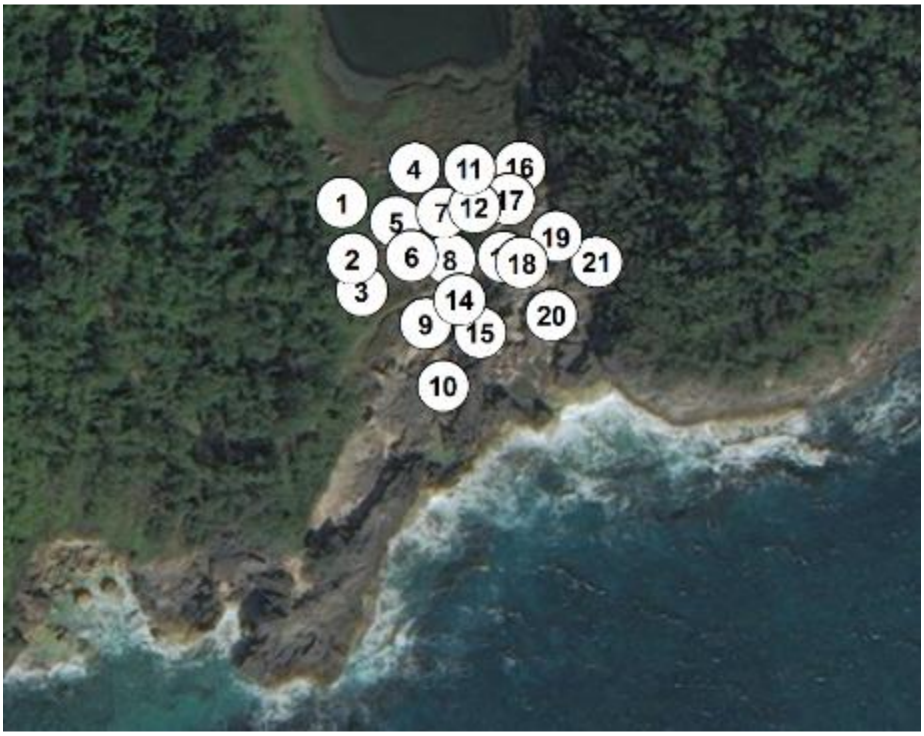


Aerial image of Inner (Inner) Pear Rock and Inner (Middle) Pear Rock showing spatial placement and numbering of traps (23 traps set).

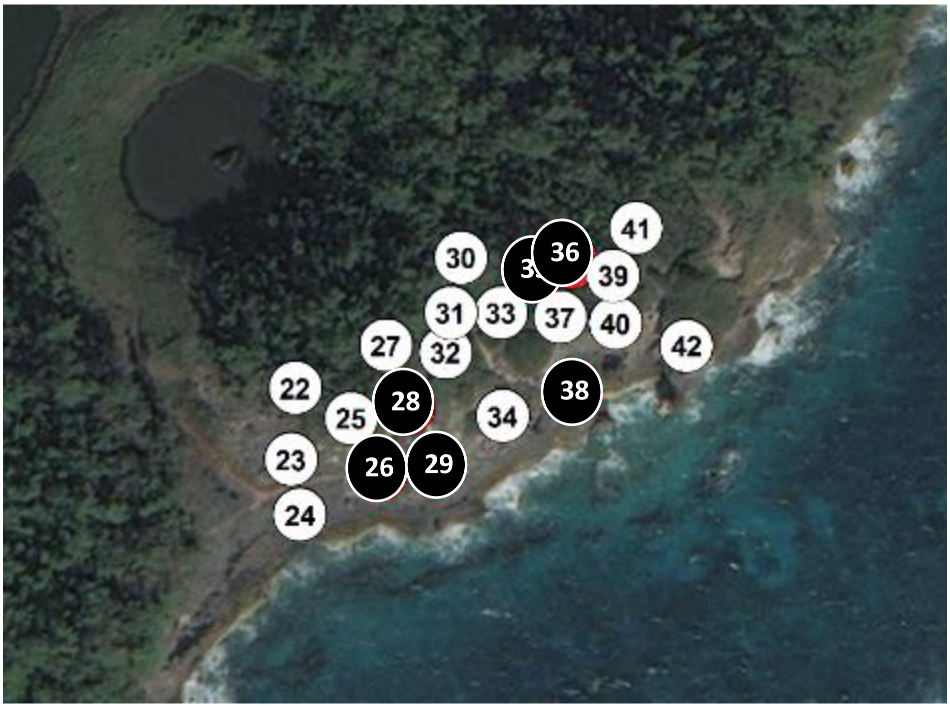
Aerial image of Castle Island showing spatial placement and numbering of traps (65 traps set).

Aerial image of Nonsuch Island showing spatial placement and numbering of traps (72 traps set).

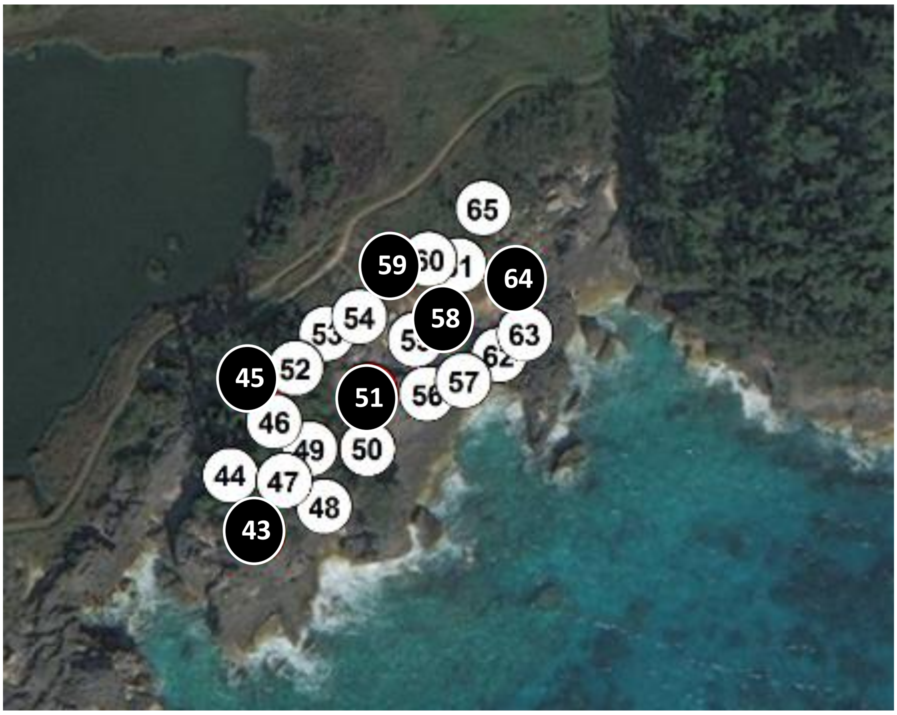
Aerial image of Spittal Pond- Checkerboard showing spatial placement and numbering of traps (21 traps set).



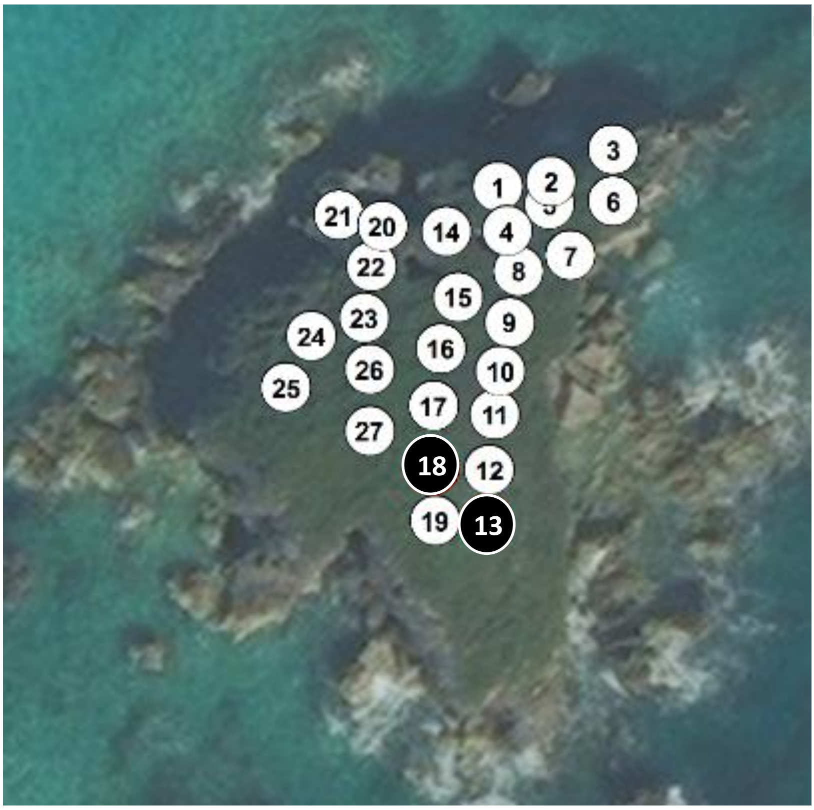
Aerial image of Spittal Pond- Jefferies Hole showing spatial placement and numbering of traps (21 traps set).



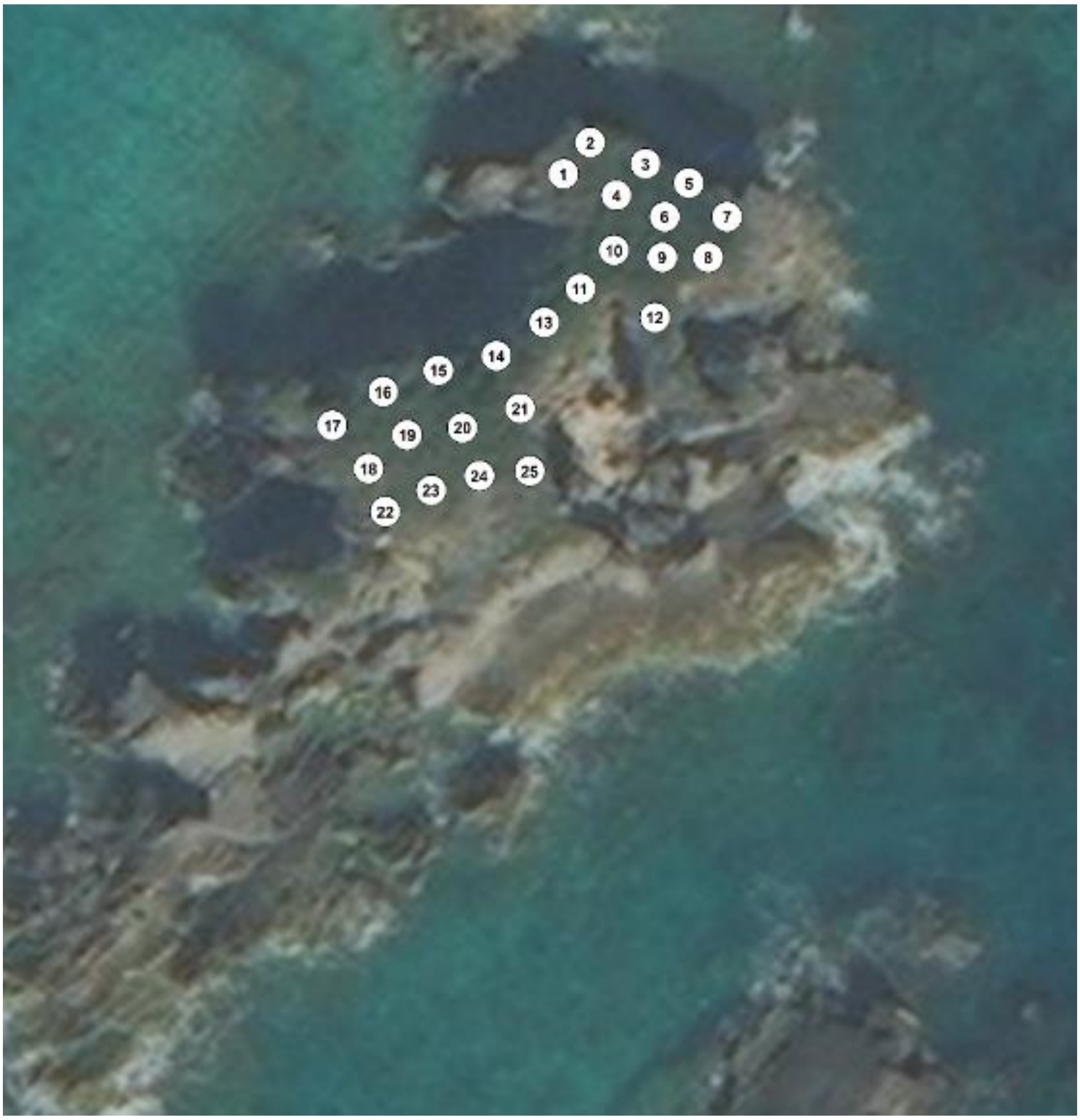
Aerial image of Spittal Pond- East End showing spatial placement and numbering of traps (23 traps set).



Aerial image ps (27 traps set).



Aerial image of Outer Pear Rock showing spatial placement and numbering of traps (25 traps set).



Aerial image of Green Rock showing spatial placement and numbering of traps (30 traps set).

Aerial image of John Smiths Park showing spatial placement and numbering of traps (30 traps set).

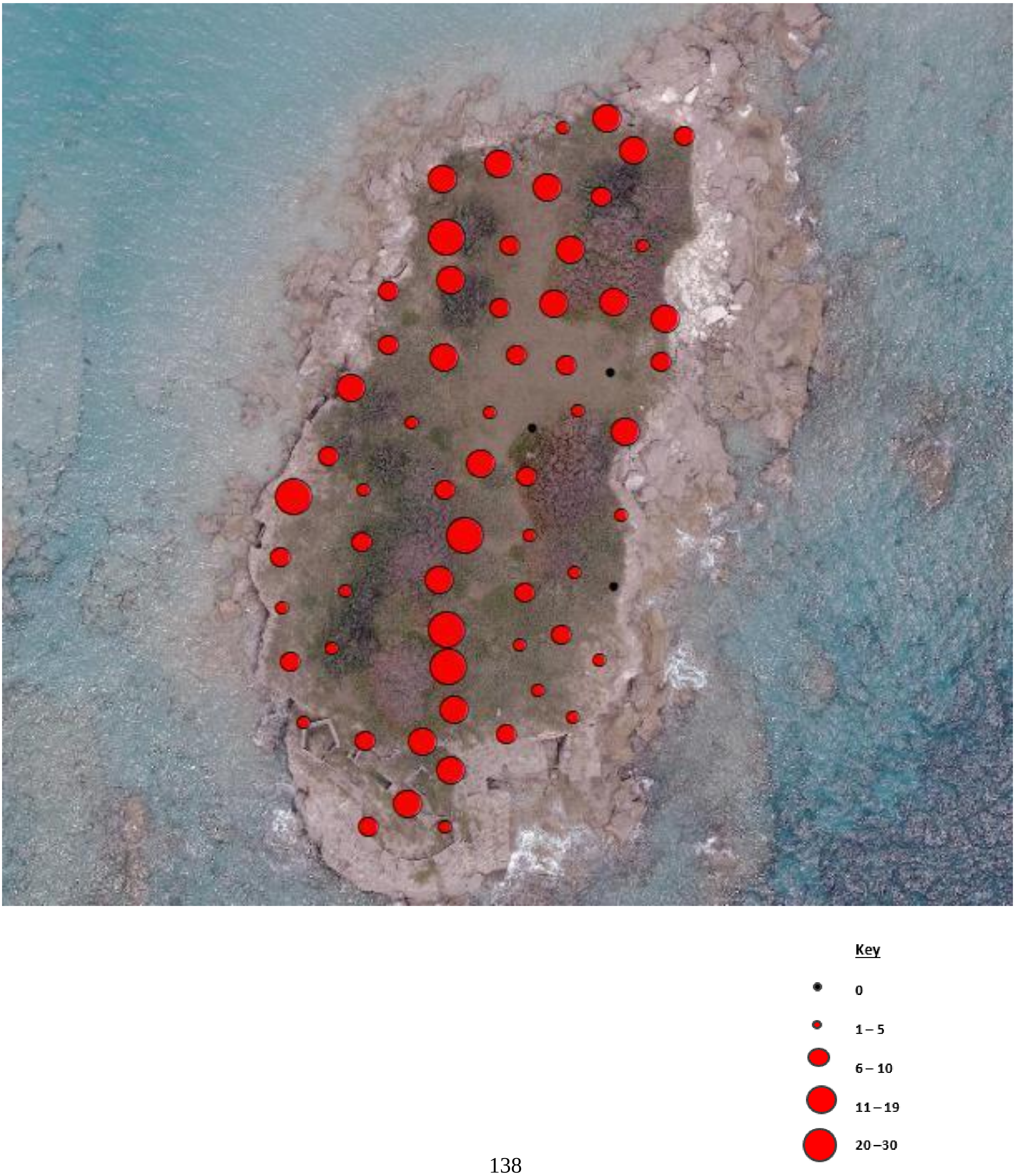
Aerial image of Church Bay Park showing spatial placement and numbering of traps (30 traps set).

Aerial image of Great Head Park showing spatial placement and numbering of traps (40 traps set).

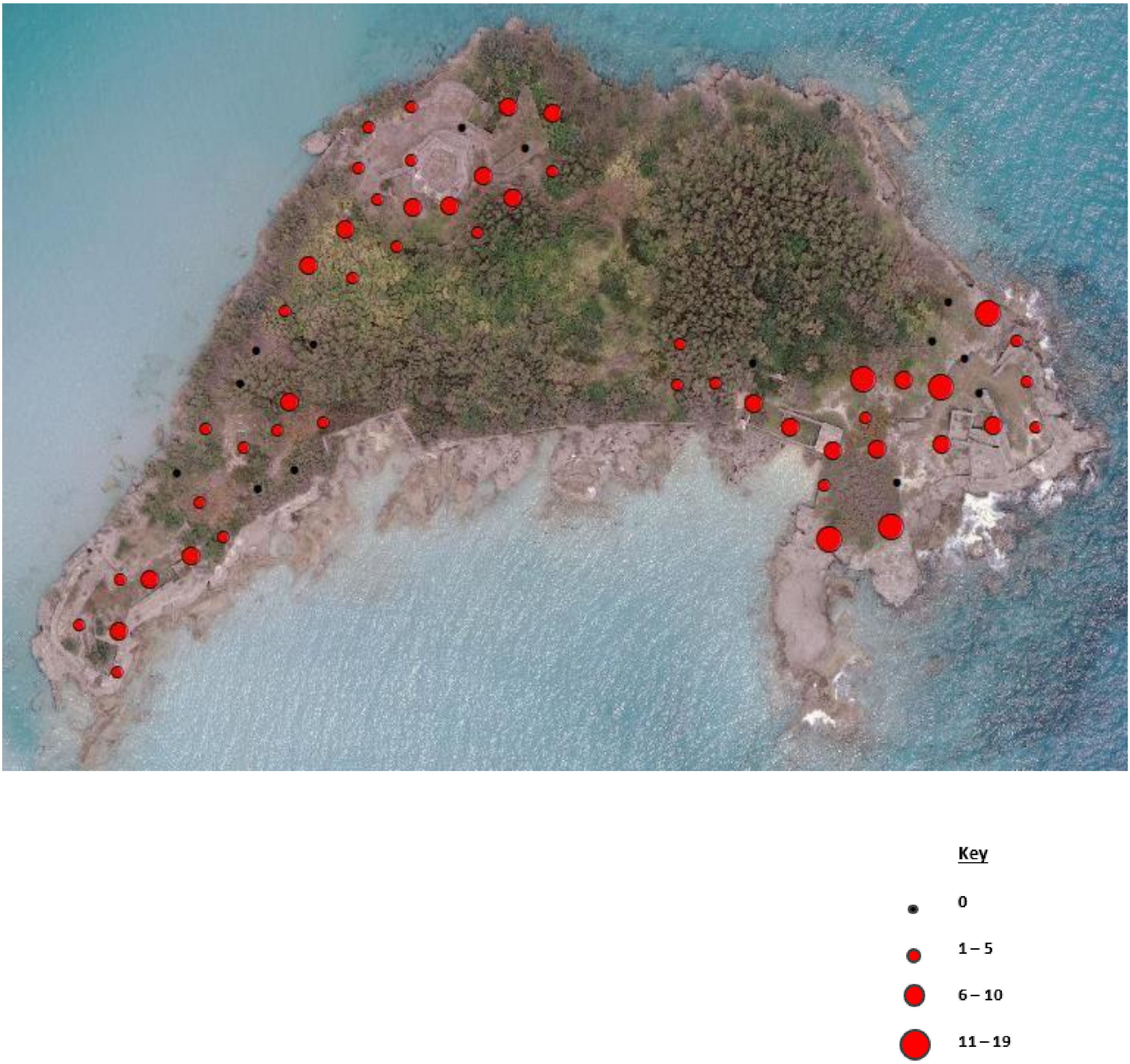
Aerial image of Warwick long bay showing spatial placement and numbering of traps (30 traps set).

## APPENDIX 3C – Frequency of skink captures on Southampton Island, Castle Island, Nonsuch Island and Spittal Pond between 2015 and 2017.

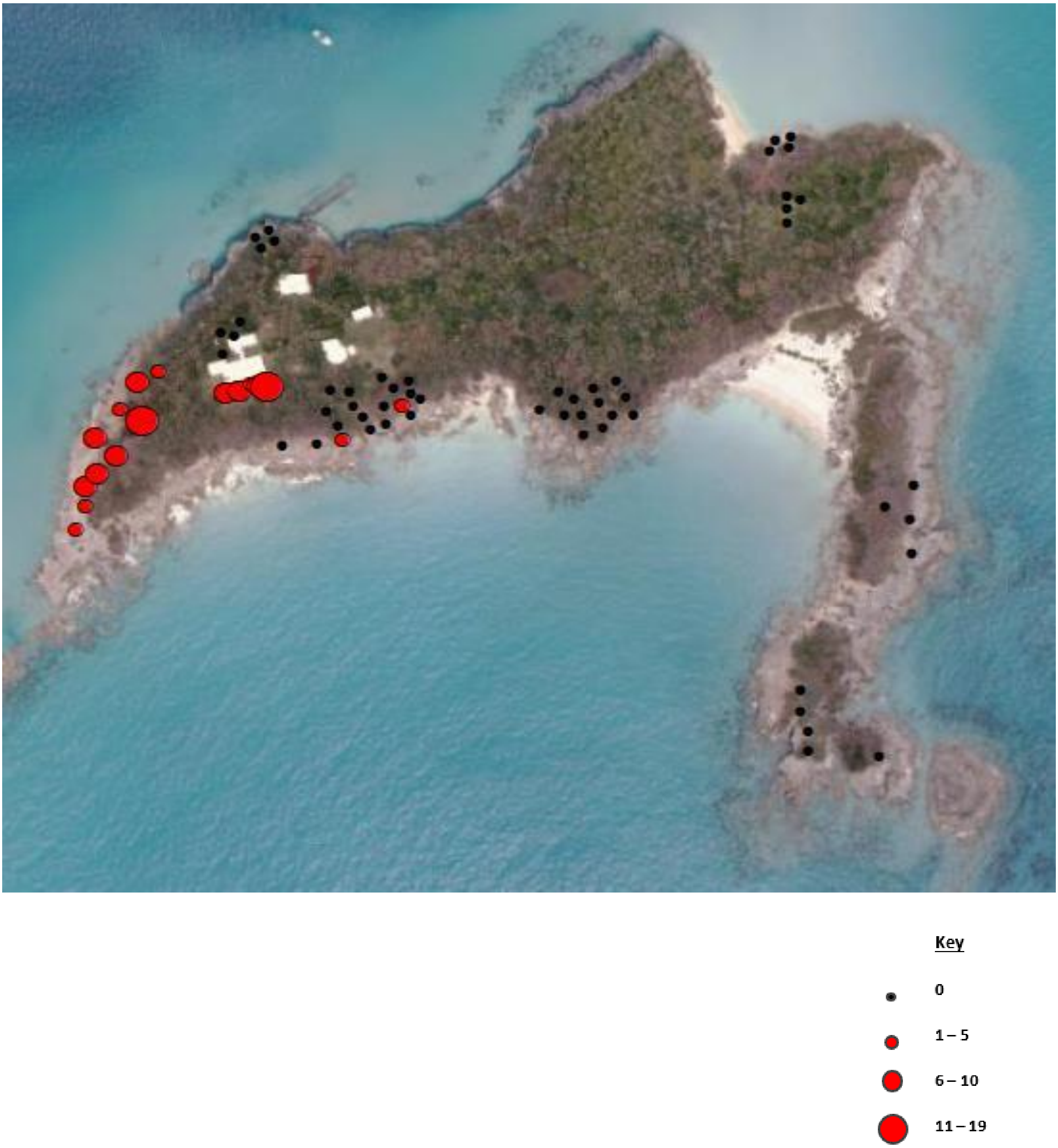
Frequency of skinks on Southampton Island.



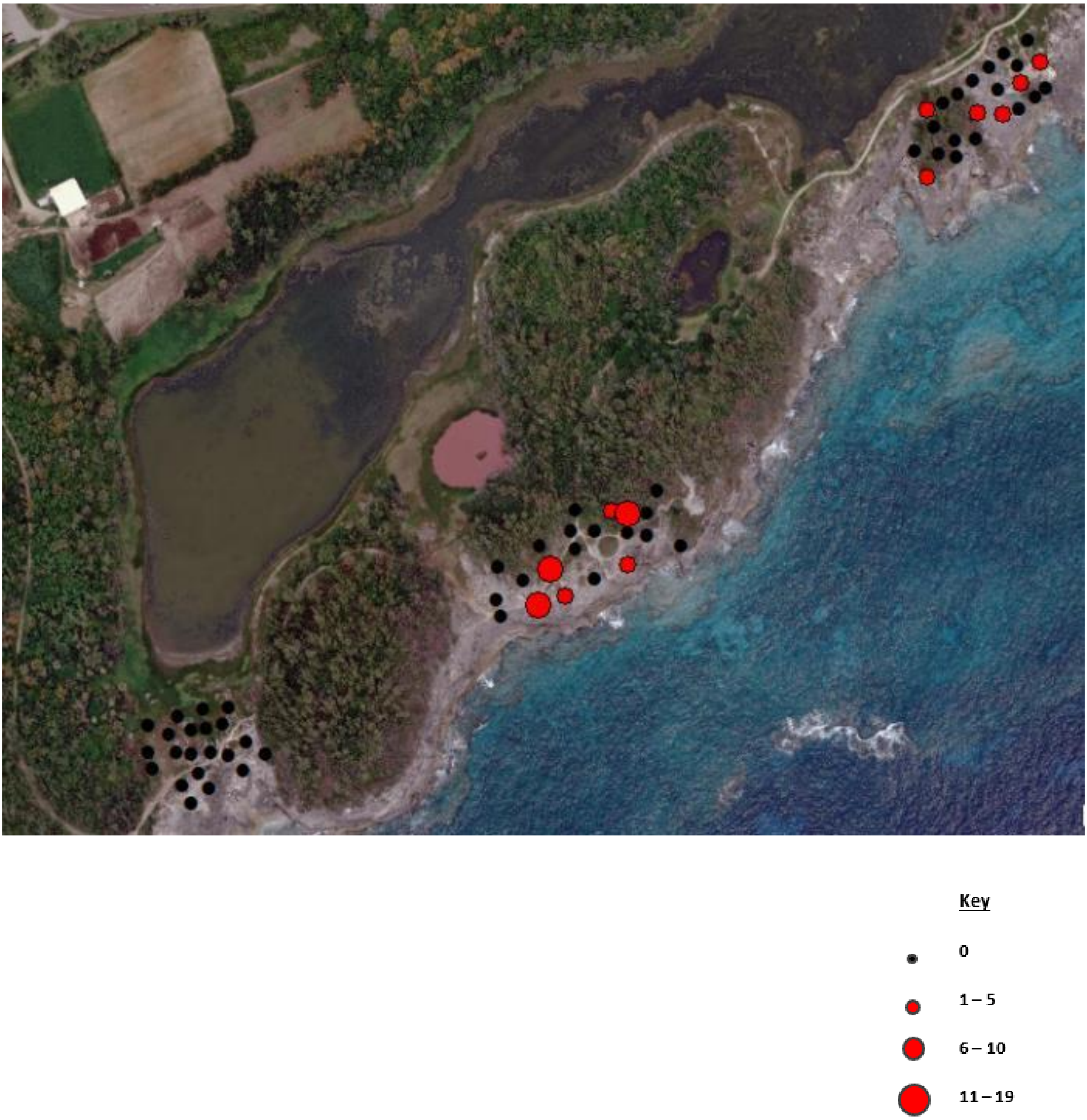
Frequency of skink captures on Castle Island.



Frequency of skink captures on Nonsuch Island.



Frequency of skink captures at Spittal Pond.



# CHAPTER 4

# VARIATION IN BODY CONDITION OF BERMUDA SKINKS IN SPACE AND TIME: IMPLICATIONS FOR HABITAT CHANGE AND CONSERVATION.

HELENA TURNER AND RICHARD A. GRIFFITHS

An adult Bermuda Skink (*Plestiodon longirostris*).
(*Photo credit: Labisko. J, 2016*).

## 4.1 ABSTRACT

Measuring body condition can provide a surrogate measure of an animal's fitness. Here, we compare the scaled mass index (SMI) – a body condition index based on mass–length relationships – to compare fitness between seven geographically isolated Bermuda skink (*Plestiodon longirostris*) sub-populations across Bermuda. Data were collected over two-time periods separated by at least 15 years (1997 – 2000 and 2015 – 2017). This allowed comparisons to be made over time, between sub-populations and between different habitats. Body condition varied between sub-populations, but at the overall population level, declined significantly over the past 15 – 20 years. Skinks in suboptimal forested habitats had lower body condition than those from more favourable coastal rock and scrub habitats. The two largest sub-populations – on Castle and Southampton Islands – had significantly higher body condition compared to other sub-populations. The body condition of those from the smallest fragmented rocky islets (Inner Pear Rock and Palm Island) decreased over time and currently have the poorest body conditions compared with those from other sub-populations. These patterns possibly reflect environmental stressors, such as losses of optimal and available habitat and resources, increased predation and other anthropogenic threats the skinks face.

## 4.2 INTRODUCTION

External measures of animal size such as body mass, snout-vent length (SVL), and internal measures such as fat, organ mass, lipid, water or protein content (e.g. Cavallini, 1996; Sears, 1998; MacCracken et al., 2012; Weatherhead et al., 1996) are essential indicators used to develop body condition indices (Green, 2001). Body condition indices can be used to reveal the stresses or limitations affecting those individuals (Schulte-Hostedde et al., 2001) and can be an indicator of an animal's foraging success (Blanckenhorn, 1991), ability to cope with competition (Strain and Johnson, 2009), environmental pressures (Carrascal et al., 1998), mating behaviour (Shine et al., 2003), or nutrition (Eley et al., 1989). However, some methods to measure body condition can be invasive, time-consuming and difficult to implement in the field (Kotiaho, 1999; Marshall et al., 1999). Indeed, invasive measures can even be fatal for the animal (Schulte-Hostedde et al., 2005).

Body condition can be measured using a range of different indices, most of which involve a measure of body mass that controls for variation in size. For example, a simple method is the residual index, which uses the residuals from an ordinary least squares (OLS) regression of body mass against one or more length measurements (Sokal and Rohlf, 1995; Bancila et al., 2010; Cattet et al., 2002). However, some indices may yield misleading information about body condition. For example, the cube root of mass divided by length may in fact reflect variation in size rather than condition (Bradshaw and De'ath, 1991), and may not account for changes in body proportions during growth (Dudek et al., 2015). Consequently, the choice of index can influence results (Jakob et al., 1996).

As previously demonstrated in amphibians, birds and mammals (Peig and Green, 2009, 2010; MacCracken and Stebbings, 2012), the scaled mass index (hereafter SMI) is a useful tool that may perform better than other alternative methods as a measure of body condition (Peig and Green, 2009). So far, the SMI has not yet been adopted in scincid lizards. A major benefit of the SMI (defined formally in the Methods section below) is that it is based on the scaling relationship between body mass and length, allowing all individuals to be at the same growth phase, so that the changing relationship with body size can be accounted for (Peig and Green, 2010).

The Bermuda skink (*Plestiodon longirostris*) is an endemic species that was once widespread across Bermuda. However, populations have been declining since the 1970s (Davenport et al., 2001), and they are now classed as critically endangered since only a few isolated sub-populations remain. The main threats include anthropogenic disturbances, coastal developments, habitat loss and the introduction of invasive flora and fauna (Raine, 1998; Wingate, 1998). Despite conservation concerns, detailed life history data is difficult to obtain because of the cryptic behaviour of the species.

The aim of this study was to calculate body condition indices using a scaled mass index (SMI) to compare Bermuda skink body condition between seven sub-populations. We predict that skinks living in areas of degraded forest habitat will have a lower SMI than those in less disturbed coastal habitats, and that SMI may have changed over time as habitats have become more degraded and disturbed. Understanding how these factors shape an animal's condition can be used as a novel indicator of ecosystem health to help develop effective management strategies.

## 4.3 METHODOLOGY

### *4.3.1 Study area*

Skinks were sampled from seven of the largest sub-populations across Bermuda: Palm Island, Spittal Pond, Castle Island, Charles Island, Southampton Island, Nonsuch Island, and Inner Pear Rock (Fig. 4.1). Data from earlier surveys were made available by the Government of Bermuda, Department of Environment and Natural Resources. Data from surveys on Palm Island and Castle Island were collated from Wingate (1998); data from surveys at Spittal Pond, Inner Pear Rock and Charles Island were collated from Raine (1998). To increase the sample size from Castle Island additional data from Hammond (2000) were also included. Although surveys were undertaken on Nonsuch Island by Griffith et al. (1993) and on Southampton Island by Davenport et al. (1997) and Glasspool and Outerbridge (2004), body mass data were not collected so lizard body condition could not be estimated. Data from 'present' surveys were collected between 2015 and 2017. Although Bermuda skinks are active all year round (Edgar et al., 2010) all population monitoring was carried out between June and September when skinks are most active (Raine, 1998).

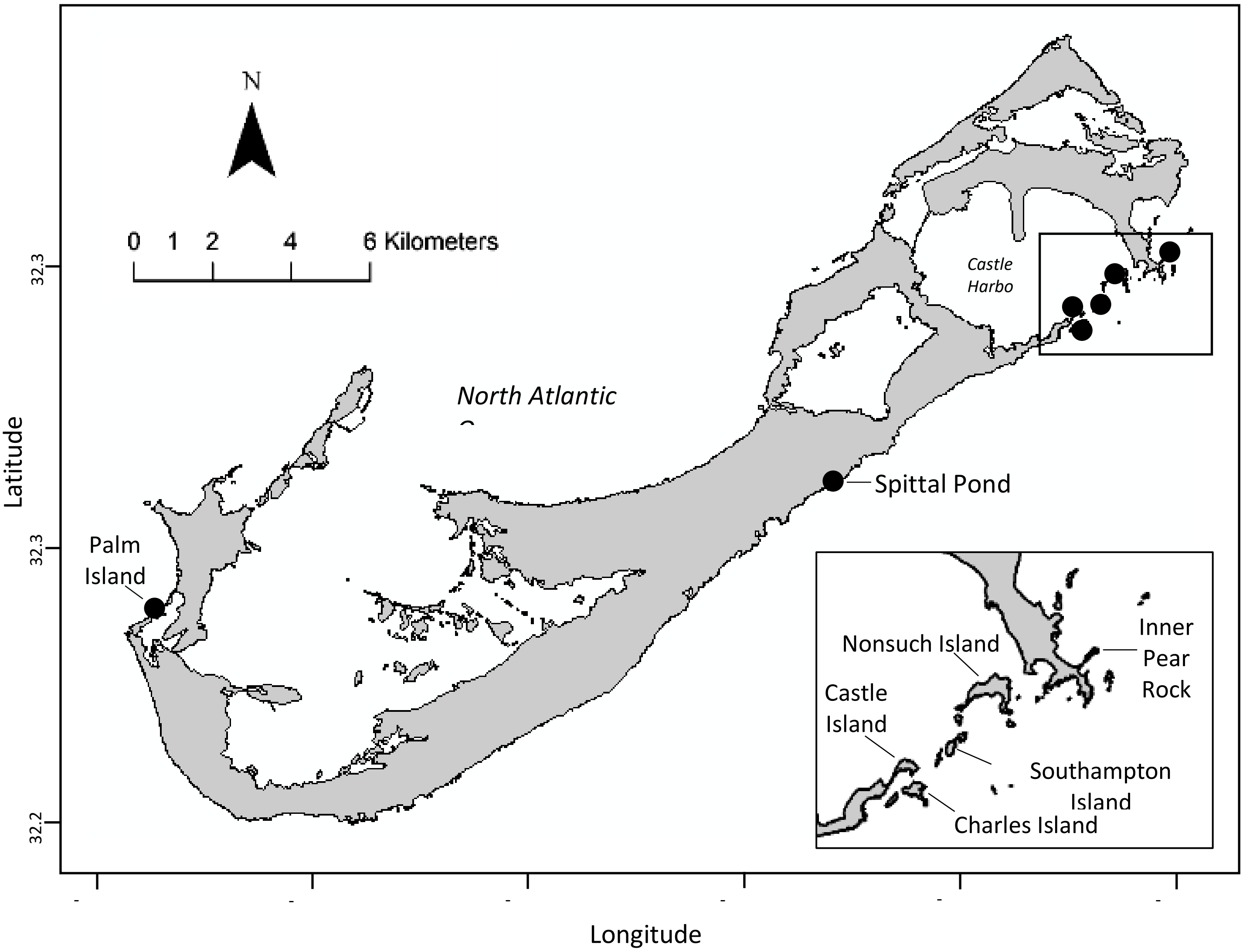


**FIGURE 4.1**. Map showing seven localities from which morphometric data were collected across Bermuda between 1997 and 2017. (Images created in ArcMap 10.5).

### *4.3.2 Live capture and handling*

The methodology was consistent between surveys and followed Davenport et al. (1997) where individuals were captured using glass jar (4.5 litres) pit fall traps and Bumblebee® sardines in oil as bait. Traps were placed approximately every 5 – 10 metres, as this is considered to embrace different home ranges (Raine, 1998). Each trap was placed at a 45° angle against a rock to prevent escapes and covered with a large palmetto leaf or damp towel to prevent the traps overheating. The number of traps set depended on the size of the site and varied between 30 and 104. Traps were checked hourly between 1100 and 1600 hrs.

Individuals captured in 'Present' surveys (2015 – 2017) were recognised individually by having their passive integrated transponder (PIT) tag scanned (see Chapter 2). For individuals surveyed in the ‘Past’ studies (1997 – 2000) or those that were too small to be tagged (< 11.0 g), a spot of coloured paint was applied to either the limbs or dorsal surface that would be detectable for the duration of the study. Lizards that had been captured already were immediately released from the trap they were caught.

A combination of body scale colouration and morphological measurements were used to classify specimens into four age classes: hatchlings, juveniles, sub-adults or adults (Davenport et al., 1997; Raine, 1998; Wingate, 1998; Glasspool and Outerbridge, 2004; Appendix 4A). Any scars or injuries to the body, limb or tail were also recorded for each individual.

### *4.3.3 Measurement of body size and mass*

All skinks collected from the traps were immediately placed in a polythene bag and weighed using a 60 g Microline Pesola® spring balance (precision: ± 0.3 g) to record individual body mass (g). Snout-vent length (SVL) was measured with a ruler (to the nearest 1.0 mm) that provided a flat surface to support the lizard, so they could be measured accurately and reduced the number of escapes. Once processed, all lizards were released at the point of capture.

### *4.3.4 Calculation of indices and statistical analyses*

We used the scaled mass index (SMI) to assess skink body condition, as recommended by Peig and Green (2009). The SMI accounts for normal growth processes (i.e. scaling) between body mass (M) and length measurements (L). It therefore allows comparisons to be made between individuals from the same species but differing in body size by standardising all individuals to the same growth phase (Lleonart et al., 2000).

$$\mathrm{SMI_i} = \widehat{M}_\mathrm{i} \times (\frac{\mathrm{L_0}}{\mathrm{L_i}})^\mathrm{b}$$

Where $\widehat{M}_i$ and $L_i$ are, respectively, the body mass and the snout-vent length of the $i^{\text{th}}$ individual; and $L_0$, is mean value of snout-vent length for the study population. The calculation of *b,* known as the ‘allometric or scaling exponent’, was the slope estimate of a standardised major axis (SMA) regression for all lizards within each sub-population (*b* values shown in Appendix 4B).

After verifying that residuals of SMI were normally distributed (Kolmogorov-Smirnov test) and had homoscedastic variance, the resulting SMI indices were used to assess trends in skink body condition by using a linear mixed-effects model (LMM). Models were fitted and compared using a restricted maximum likelihood estimation method. We used SMI as the dependent variable and included site, habitat and age class as dependent fixed-effects, and the time (past and present), as an independent random effect. Upon finding a significant effect of time on skink body condition, we explored temporal trends in body condition between habitat and age classes. Body condition could only be compared between four of the seven sub-populations as no ‘Present’ data were available for Charles Island and no ‘Past’ data for Southampton Island or Nonsuch Island.

Post-hoc pairwise comparisons of SMI were based on estimated marginal means and were Bonferroni corrected (Rice, 1989). Comparisons of the mutilation rate between ‘past’ (1997 – 2000) and ‘present’ (2015 – 2017) were conducted using the "N-1" Chi-squared test as recommended by Campbell (2007) for small sample sizes.

The level of significance was set at 0.05 and all computations unless stated otherwise were performed using the ‘lmer’ function in the ‘lme4’ package in R software (version 3.4.3, R Core Team, 2017).

## 4.4 RESULTS

### *4.4.1 Comparisons between sub-populations*

A total of 921lizard measurements were collected from seven Bermuda skink sub-populations: Southampton Island, Castle Island, Nonsuch Island, Spittal Pond, Inner Pear Rock, Charles Island and Palm Island between 1997 and2017(Table 4.1). There were significant differences in body condition between sites (LMM: $F(6,912) = 6.11$, $P < 0.01$; Fig. 4.2). Post-hoc analysis showed three significant groupings ($P < 0.01$): A) Southampton Island and Castle Island B) Charles Island, Nonsuch Island and Spittal Pond and C) Palm Island, and Inner Pear Rock. Group A had the greatest SMI, whereas group C had the lowest SMI.

**TABLE 4.1.**

Sample sizes (*N*), mean snout-vent length (SVL, mm), mass (g), scaled mass indices (SMI) and standard errors (SE) for seven Bermuda skink sub-populations. Both 'Past' and 'Present' data (1997 – 2017) are combined.

| Site | *N* | Body mass ± SE | SVL ± SE | SMI ± SE |
|---|---|---|---|---|
| Charles Island | 62 | 16.30 ± 0.53 | 80.70 ± 1.12 | 15.98 ± 0.16 |
| Castle Island | 249 | 17.12 ± 0.28 | 76.61 ± 0.54 | 17.21 ± 0.10 |
| Inner Pear Rock | 41 | 9.00 ± 0.46 | 64.47 ± 1.23 | 9.84 ± 0.32 |
| Nonsuch Island | 49 | 19.50 ± 0.90 | 79.00 ± 0.87 | 15.92 ± 0.35 |
| Palm Island | 30 | 13.11 ± 0.71 | 78.20 ± 1.37 | 12.51 ± 0.20 |
| Southampton Island | 406 | 20.31 ± 0.27 | 81.86 ± 0.46 | 17.46 ± 0.10 |
| Spittal Pond | 84 | 15.44 ± 0.50 | 75.69 ± 0.81 | 15.08 ± 0.22 |
| Total | 921 | 15.83 ± 0.52 | 76.65 ± 0.91 | 14.86 ± 0.21 |

### *4.4.2 Changes in body condition over time*

Associations were evident (Fig. 4.2) between body condition and habitat type (LMM: $F(2,916) = 9.55$, $P < 0.05$). Skinks caught in scrub habitats ($n = 507$, mean SMI = 17.57 ± 0.57) had a higher mean SMI than those caught in rock ($n = 330$, mean SMI = 14.76 ± 0.42) or forest habitats ($n = 84$; mean SMI = 12.91 ± 0.36).

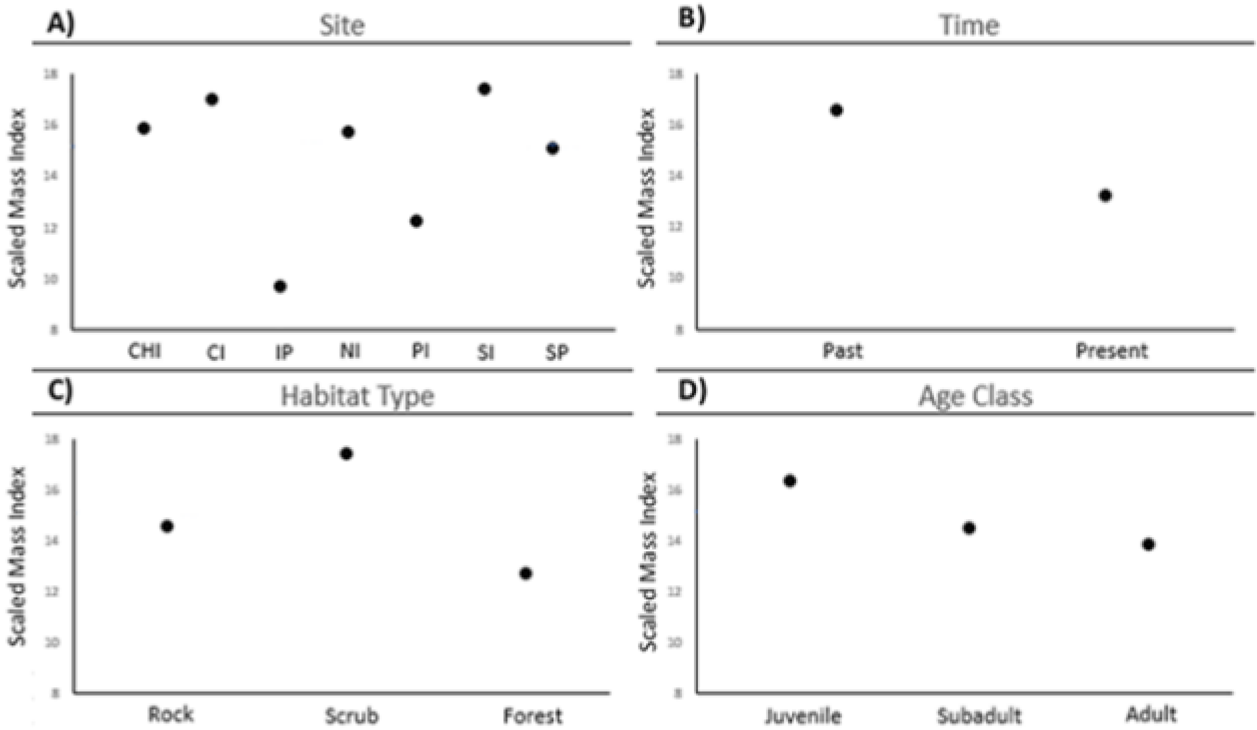


**FIGURE 4.2**. Mean scaled mass index (SMI) between A) site B) time C) habitat type and D) age classes. Data collected from seven Bermuda skink sub-populations (CHI = Charles Island, CI = Castle Island, IP = Inner Pear Rock, NI = Nonsuch Island, PI = Palm Island, SI = Southampton Island, SP = Spittal Pond) between 1997 and 2017.

Interactions between time and habitat type showed that those caught in the past from scrub habitat (*n* = 157, mean SMI = 19.86 ± 0.29) had higher SMI compared to those caught in the past, in rock habitat (*n* = 172, mean SMI = 15.98 ± 0.55) or forest habitat (*n* = 75, mean SMI = 14.36 ± 0.49, LMM: *F*(2,397) = 10.02, *P* < 0.01). Comparing the SMI between those caught in the past and present, the SMI of those caught in rock habitat declined by 15.5%, those caught in scrub habitat declined by 22.9% and those caught in forest habitat declined by 20.4% (Fig. 4.3).

Although there was no significant interaction between body condition and age classes (LMM: *F*(2,916) = 0.21, *P* = 0.813), there was an interaction between time and age class (LMM: *F*(2,397) = 4.29, *P* < 0.05). Over time, juvenile body condition declined by 17.3%, sub-adults by 17.5 % and adults by 24.0% (Fig. 4.3).

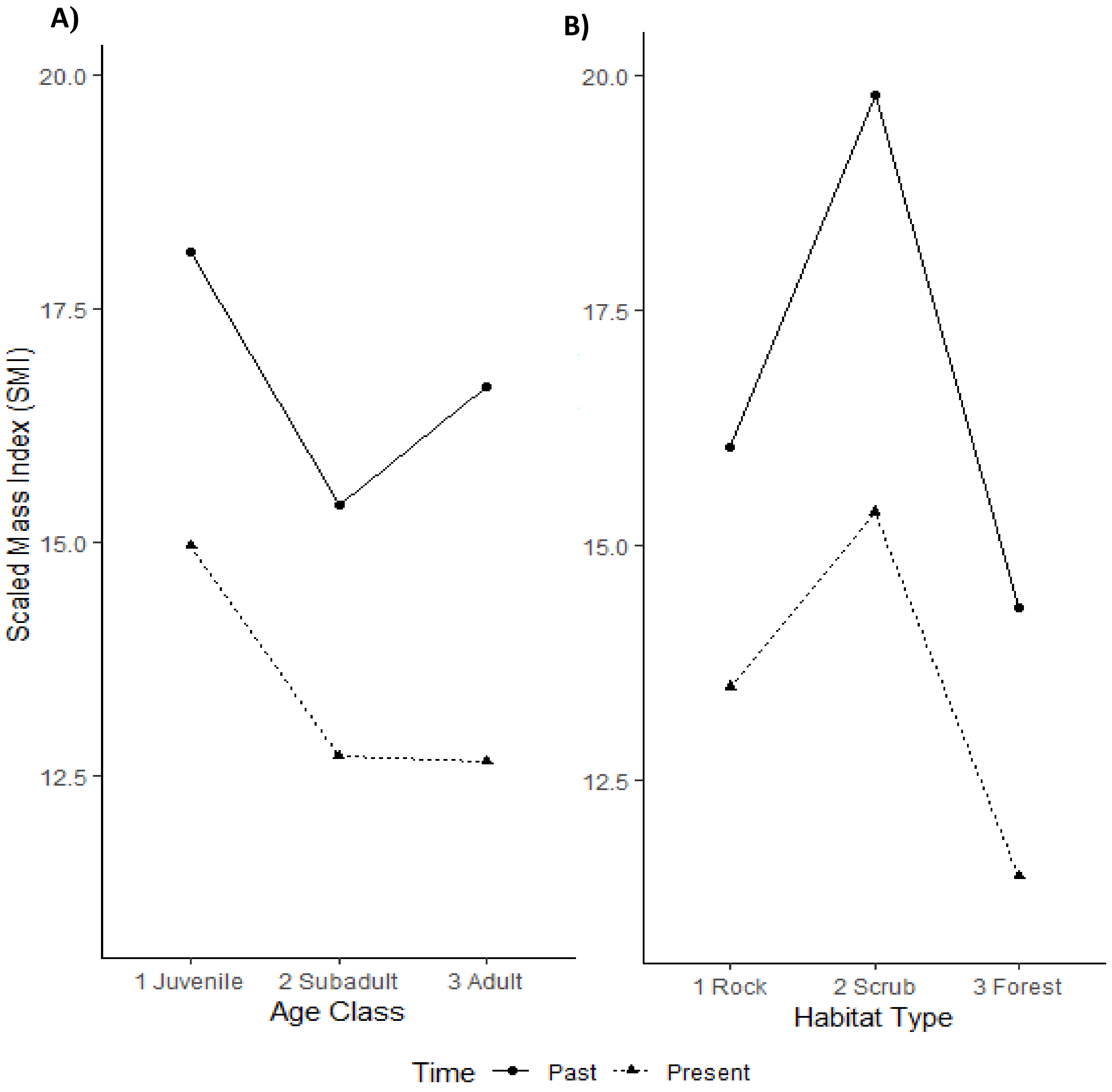


**FIGURE 4.3**. Mean scaled mass index (SMI) interactions between A) time (Past and Present) and age classes (Juvenile, Sub-adult, Adult) and between B) time (Past and Present) and habitat type (Rock, Scrub, Forest).

The random effects model showed body condition significantly decreased at Palm Island and Inner Pear Rock over time (Table 4.2). Overall, results showed there was a 19.6% decline in SMI over time (LMM: $F(1,917) = 13.91$, $P < 0.05$) between past sites ($n = 169$, mean SMI = 16.72 ± 0.45) and present sites ($n = 230$, mean SMI = 13.44 ± 0.32).

**TABLE 4.2.**

Results of random effects model of body condition comparisons between four of the seven Bermuda skink sub-populations over the periods 1997 – 2000 and 2015 – 2017. Direction of t-values (positive or negative) indicates the direction of the linear trend over time.

| Site | *t* | df | *P-Value* |
|---|---|---|---|
| Castle Island | 2.677 | 6.36 | 0.495 |
| Inner Pear Rock | -2.422 | 8.19 | < 0.05* |
| Palm Island | -3.004 | 8.04 | < 0.05* |
| Spittal Pond | -1.781 | 8.93 | 0.089 |

### *4.4.3 Mutilation rate*

More than half of the skinks caught in both the past and the present time periods exhibited some form of mutilation. There was a significant relationship between the number of individuals that had tail or toe mutilations between the past and the present ($\chi^2 = 80.47$, df = 1, $P < 0.001$). Tail loss increased by 32.4% over time compared to toe loss which has decreased by 9.3% over time (Table 4.3).

**TABLE 4.3.**

Percentage (%) of mutilation among skinks captured at each site in the 'Past' surveys between 1997 – 2000 and the 'Present' surveys between 2015 – 2017 and mean percentages used to compare samples. A hyphen (-) indicates insufficient data available.

| Site | Past | | Present | |
|---|---|---|---|---|
| | Tail Loss | Toe Loss | Tail Loss | Toe Loss |
| Charles Island | 61 (38/62) | 10 (6/62) | - | - |
| Castle Island | 50 (39/78) | 21 (16/78) | 88 (150/170) | 3 (5/170) |
| Inner Pear Rock | 60 (22/37) | 11 (4/37) | 100 (2/2) | 0 (0/2) |
| Nonsuch Island | - | - | 98 (45/46) | 4 (2/46) |
| Palm Island | 63 (19/30) | 10 (3/30) | 100 (1/1) | 0 (0/1) |
| Southampton Island | - | - | 86 (258/299) | 4 (13/299) |
| Spittal Pond | 73 (45/62) | 8 (5/62) | 91 (20/22) | 5 (1/22) |
| *Mean percentage* | 61.4 % | 12.0 % | 93.8 % | 2.67 % |

Past survey data collected by: Raine (1998), Wingate (1998) and Hammond (2000).

## 4.5 DISCUSSION

Using the SMI provides a simple, reliable, non-invasive procedure to compare the body condition between conspecifics regardless of age, reproductive state, geographical population, or date of capture. Inner Pear Rock and Palm Island were found to have the lowest body condition indices. Both are small (< 0.4 hectares) inhospitable offshore islands. Differences in SMI may be due to availability of resources, especially as the two largest sites, Southampton and Castle Island, had the highest SMI in comparison. The body condition of skinks on Castle Island increased over time, whereas it declined on Inner Pear Rock and Palm Island. As Inner Pear Rock and Palm Island are the smallest sites surveyed, they only have a very small number of seabirds that nest annually. Consequently, the supply of dropped fish or broken eggs from nesting seabirds may not always be available for the skinks. Although prickly pears are found on both islands, their fruits are also only seasonally available, again reducing the amount of available resources. Body condition has been found to deteriorate in Marine iguana (*Amblyrhynchus cristatus*) populations in the Galapagos as a result of changing environmental conditions. The largest animals (males) on each island were the first to die once environmental conditions deteriorated especially during El Niño events (Wikelski and Trillmich, 1997; Wikelski and Thom, 2000). Possibly there is size-related mortality amongst Bermuda skinks living in poorer quality environments. However, there is much stronger sexual size dimorphism in marine iguanas compared to Bermuda skinks.

Individuals caught in forest habitat had the lowest mean body condition compared to those caught in rock or scrub habitat. Although Bermuda skinks are most commonly found within rocky coastal habitat (Glasspool and Outerbridge, 2004), the islands within Castle Harbour have increasing amounts of non-native vegetation such as dense stands of asparagus fern (*Asparagus densiflorus*)*,* Brazil pepper (*Schinus terebinthifolia*) and casuarina trees (*Casuarina equisetifolia;* Bernews, 2015). There has been a 26.2% increase in the resident human population in Bermuda in the past 50 years (50,520 in 1960 to 63,779 in 2010; Government of Bermuda, 1960; Government of Bermuda, 2016) and suitable native habitat is slowly being lost across Bermuda (Griffith et al., 1993). Habitat has changed particularly on the coastlines from a predominantly tussocked crabgrass (*Stenotaphrum secundatum*) habitat, to a more complex mixed forest, more suitable for invasive species such as Kiskadee flycatchers

*Pitangus sulphuratus*, yellow-crowned night herons *Nycticorax violacea,* brown and black rats *Rattus* sp. and Jamaican anoles *Anolis grahami* known to predate the skinks (Griffith et al., 1993). These forested habitats are unsuitable for the skinks because they reduce opportunities to forage, bask and nest. Several studies have shown that invasive plants generate changes to the structure and composition of original communities, reducing species richness and abundance of native plants (Higgins et al., 1999; Ferdinands et al., 2005), invertebrates (Herrera and Dudley, 2003), and vertebrate populations (Meik et al., 2002). Introduced plants may even alter localised environmental conditions by changing the temperature and humidity (Marchante et al., 2008). The introduction of invasive vegetation would also create unfavourable nesting conditions to native seabirds. As a result, birds may have problems accessing nests which may limit breeding opportunities and therefore reduce the amount of available resources (e.g. dropped fish and broken eggs) to the skinks.

The Bermuda skink population is clearly in decline, with no skinks seen at Charles Island, only one skink recorded at Palm Island and numbers on the mainland - especially at Spittal Pond - declining fast. This is particularly worrying considering that the earlier surveys in 1998 suggested the populations were healthy (Raine, 1998; Wingate, 1998). It is likely that a combination of suboptimal habitat and high predation levels is influencing the fitness of the skinks, and ultimately, population size.

Juveniles in poorest body condition were caught at Nonsuch Island and Spittal Pond. Although fewer juveniles were caught at these two sites compared to the other sub-populations, previous studies found Nonsuch Island and Spittal pond to have ageing populations and poor recruitment into the breeding population (Griffith et al., 1993; Raine, 1998; Wingate, 1998). This may lead to reduced population viability to habitat quality, predation levels or low fertility due to inbreeding.

Although the single largest individual was caught on Nonsuch island (32 g), mean body condition at Spittal pond and Nonsuch Island has declined by an average 7.28% over the past twenty years. During surveys undertaken between 2015 and 2017, Bermuda experienced an El Niño event, which resulted in milder weather. Such environmental fluctuations can result in seasonal changes in food availability which many studies on reptiles have shown feeding success correlates with growth rates (e.g. Jakob and Dingle, 1990; Passos et al., 2014) and egg production (e.g. Morse, 1988), and could also temporarily affect the lizards' ability to bask and undertake natural foraging and breeding behaviours, which ultimately affects their future fitness.

There was a significant difference between tail and toe mutilations over time. Tail loss has been found to reduce mating success, reduce activity levels and affect locomotion in lizards, so may therefore decrease the fitness of individuals (Vitt, 1983; Martin and Salvador, 1995, 1997; Salvador et al., 1995; Martin and Avery, 1998). However, if mutilation is related to high rates of predator-inflicted injuries, this could reflect predator inefficiency rather than predation pressure (Schoener, 1979; Arnold, 1988). High frequencies of toe loss in lizards are unlikely to have a severe effect on survival (Bloch and Irschick, 2004). They are often an indication of intraspecific competition (e.g. Chadwick, 1991; Soederbaeck, 1995) or aggression (e.g. Kramer, 1951; Blair, 1960; Tinkle, 1967; Pianka, 1973), especially in high density populations (Vervust et al., 2009). However, as toe mutilations decreased over time this may indicate less intraspecific competition for resources as skink populations become less abundant.

### *4.5.1 Conclusion*

Monitoring environmental conditions and microhabitats alongside the body condition of Bermuda skinks may provide early warning signs to detect fluctuations that may negatively impact survival. Although body condition is just a surrogate index of potential pressures on populations, this study provides important information for conservation managers. Further studies should compare the response of SMI to specific health indicators, such as parasite load, growth rates, diet quality, survivorship and reproductive success that are likely to influence the variation in individual fitness. Likewise, the use of non-invasive methods (i.e. collection of faecal samples; buccal swabs; visual health checks; behavioural observations) may generate data that could usefully be explored in relation to body condition. Although it is acknowledged that sample sizes may be smaller than working with more abundant species, this information will be valuable to further understand pressures on endangered species such as the Bermuda skink. Lastly, with a captive breeding programme for the Bermuda skinks underway, past and present body condition data from the field will provide an informative baseline for monitoring ex situ populations.

## 4.6 LITERATURE CITED

Arnold, E.N. (1988). *Caudal autotomy as a defense.* In: Gans C., and Huey RB (Eds.) Biology of the reptilia, Vol. 16. Alan R. Liss: New York, United States. Pp 235–273.

Bancila, R.J., Hartel, T., Plaiasu, R., Smets, J., and Cogalniceanu, D. (2010). Comparing three body condition indices in amphibians: a case study of yellow-bellied toad *Bombina variegata. Amphibia-Reptilia,* 31:558–562.

Bernews. (2015). *18,783 Invasive plants removed From Nonsuch*. [Online]. Available at: http://bernews.com/2015/03/nonsuch-invasive/ [Assessed 4 February 2019].

Blanckenhorn, W.V. (1991). Fitness consequences of foraging success in water striders (*Gerris remigis*; Heteroptera: Gerridae). *Behavioral Ecology,* 2:46–55.

Blair, W.F. (1960). *The rusty lizard, a population study*. University of Texas Press: United States. Pp 185.

Bloch, N., and Irschick, D.J. (2004). Toe-clipping dramatically reduces clinging performance in a pad-bearing lizard (*Anolis carolinensis*). *Journal of Herpetology*, 37:293–298.

Bradshaw, S.D., and De'ath, G. (1991). Variation in condition indices due to climatic and seasonal factors in an Australian desert lizard, *Amphibolurus muchalis*. *Australian Journal of Zoology,* 39:373–385.

Carrascal, L.M., Senar, J.C., Mozetich, I., Uribe, F., and Domenech, J. (1998). Interactions among environmental stress, body condition, nutritional status, and dominance in great tits. *The Auk,* 115:727–738.

Campbell, I. (2007). Chi-squared and Fisher-Irwin tests of two-by-two tables with small sample recommendations. *Statistics in Medicine*, 26:3661–3675.

Cattet, M.R.L., Caulkett, N.A., Obbard, M.E., and Stenhouse, G.B. (2002). A body condition index for ursids. *Canadian Journal of Zoology,* 80:1156–1161.

Cavallini, P. (1996). Comparison of body condition indices in the red fox (*Fissipedia canidae). Mammalia,* 60:449–462.

Chadwick, N.E. (1991). Spatial distribution and the effects of competition on some temperate *Scleractinia* and *Corallimorpharia*. *Marine Ecology Progress Series,* 70:39–48.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). A *study of populations of the Bermudian rock lizard (skink), Eumeces longirostris, cope (1861), on the islands of Nonsuch and Southampton, Bermuda*. Department of Agriculture and Fisheries: Bermuda.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (2001). Threats to the critically endangered endemic Bermudian skink *Eumeces longirostris*. *Oryx,* 35:332–339.

Dudek, K., Sajkowska, Z., Gawalek, M., and Ekner-Grzyb, A. (2015). Using body condition index can be an unreliable indicator of fitness: a case of sand lizard *Lacerta agilis* Linnaeus, 1758 (Sauria: Lacertidae). *Turkish Journal of Zoology,* 39:182–184.

Edgar, P., Kitson, L., and Glasspool, A.F. (2010). Recovery plan for the Bermuda skink (*Eumeces longirostris*). Government of Bermuda, Department of Conservation Services: Bermuda.

Eley, R.M., Strum, S.C., Muchemi, G., and Reid, G.D. (1989). Nutrition, body condition, activity patterns, and parasitism of free-ranging troops of olive baboons (*Papio anubis*) in Kenya. *American Journal of Primatology,* 18: 209–219.

Ferdinands, K., Beggs, K., and Whitehead, P.J. (2005). Biodiversity and invasive grass species: Multiple-use or monoculture? *Wildlife Research*, 32:447–457.

Glasspool, A., and Outerbridge, M. (2005). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island, Castle Harbour, Bermuda*. Bermuda Zoological Society: Bermuda.

Government of Bermuda. (1960). *Bermuda census 1960*. Department of Statistics. [Online]. Available at: https://www.gov.bm/1960-census [Assessed 12 February 2018].

Government of Bermuda. (2016). *2016 Population and housing census report*. Department of Statistics. [Online]. Available at: https://www.gov.bm/sites/default/files/2016%20Census%20Report.pdf [Assessed 16 September 2018].

Green, A.J. (2001). Mass/length residuals: measures of body condition or generators of spurious results? *Ecology,* 82:1473–1483.

Griffith, H., Wingate, D.B., and Robinson, D. (1993). *Introduced birds as probable agents of population changes in the Bermuda rock-lizard*. Bermuda Zoological Society: Bermuda.

Hammond, M.P. (2000). *An analysis of Bermuda Skink (Eucemes longirostris) populations on Castle Island, Bermuda*. Bermuda Zoological Society: Bermuda.

Herrera, A.M., and Dudley, T.L. (2003). Reduction of riparian arthropod abundance and diversity as a consequence of giant reed (*Arundo donax*) invasion. *Biological Invasions*, 5:167–177.

Higgins, S.I., Richardson, D.M., Cowling, R.M., and Trinder-Smith, T.H. (1999). Predicting the landscape-scale distribution of alien plants and their threat to plant diversity. *Conservation Biology*, 13:303–313.

Jakob, E.M., Marshall, S.D., and Uetz, G.W. (1996). Estimating fitness: a comparison of body condition indices. *Oikos,* 77:61–67.

Kramer, G. (1951). Body proportions of mainland and island lizards. *Evolution*, 5:193–206.

Lleonart. J., Salat, J., and Torres, G.J. (2000). Removing allometric effects of body size in morphological analysis. *Journal of Theoretical Biology*, 205:85–93.

MacCracken, J.G., and Stebbings, J.L. (2012). Test of a body condition index with amphibians. *Journal of Herpetology,* 46:346–350.

Marchante, E., Kjoller, A., Struwe, S., and Freitas, H. (2008). Short and long-term impacts of *Acacia longifolia* invasion on the below ground processes of a Mediterranean coastal dune ecosystem. *Applied Soil Ecology*, 40:210–217.

Martin, J., and Avery, R.A. (1998). Effects of tail loss on the movement patterns of the lizard, *Psammodromus algirus*. *Functional Ecology*, 12:794–802.

Martin, J., and Salvador, A. (1995). Effects of tail loss on activity patterns of rock lizards, *Lacerta monticola*. *Copeia*, 12:984–988.

Martin, J., and Salvador, A. (1997). Effects of tail loss on the time budgets, movements, and spacing patterns of Iberian rock lizards, *Lacerta monticola*. *Herpetologica*, 12:117–125.

Meik, J.M., Jeo, R.M., Mendelson, J.R., and Jenks, K.E. (2002). Effects of bush encroachment on an assemblage of diurnal lizard species in central Namibia. *Biological Conservation*, 106:29–36.

Mushinsky, H.R., Hebrard, J.J., and Vodopich, D.S. (1982). Ontogeny of water snake foraging ecology. *Ecology,* 63:1624–1629.

Passos, L.F., Coutinho, M.E., and Young. R.J. (2014). Demographic and growth analysis of broad snouted caiman (*Caiman latirostris*) in a disturbed environment in south eastern Brazil. *Herpetological Journal* 24:223–228.

Peig, J., and Green, A.J. (2009). New perspectives for estimating body condition from mass/length data: the scaled mass index as an alternative method. *Oikos,* 118:1883–1891.

Peig, J., and Green, A.J. (2010). The paradigm of body condition: a critical reappraisal of current methods based on mass and length. *Functional Ecology,* 24:1323–1332.

Pianka, E.R. (1973). The structure of lizard communities. *Annual Review of Ecology and Systematics*, 4:53–74.

R Core Team. (2017). *R: A language and environment for statistical*. R Foundation for Statistical Computing. Vienna, Austria.

Raine, A. (1998). *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the critically endangered Bermuda rock lizard (Eumeces longirostris).* MSc Dissertation. University College London: United Kingdom.

Rice, W.R. (1989). Analyzing tables of statistical tests. *Evolution*, 43:223–225.

Salvador, A., Martin, J., López, P. (1995). Tail loss reduces home range size and access to females in male lizards, *Psammodromus algirus*. *Behavioral Ecology*, 12:382–387.

Schoener, T.W. (1979). Inferring the properties of predation and other injury producing agents from injury frequencies. *Ecology*, 60:1110–1115.

Schulte-Hostedde, A.I., Zinner, B., Millar, J.S., and Hickling, G.J. (2005). Restitution of mass-size residuals: validating body condition indices. *Ecology,* 86:155–163.

Sears, J. (1988). Assessment of body condition in live birds; measurements of protein and fat reserves in the mute swan, *Cygnus olor*. *Journal of Zoology,* 216:95–308.

Shine, R., Phillips, B., Waye, H. LeMaster, M., and Mason, R.T. (2003). Chemosensory cues allow courting male garter snakes to assess body length and body condition of potential mates. *Behavioural Ecology and Sociobiology,* 54:162–166.

Soederbaeck, B. (1995). Replacement of the native crayfish *Astacus astacus* by the introduced species *Pacifastacus leniusculus* in a Swedish lake: possible causes and mechanisms. *Freshwater Biology*, 33:291–304.

Sokal, R.R., and Rohlf, F.J. (1995). Biometry. The principles and practice of statistics in biological research. Third Edition. Freeman: New York, United States.

Strain, E.M.A., and Johnson, C.R. (2009). Competition between an invasive urchin and commercially fished abalone: effect on body condition, reproduction and survivorship. *Marine Ecology Progress Series*, 377:169–182.

Tinkle, D.W. (1967). The life and demography of the side-blotched lizard, *Uta stansburiana*. *Miscellaneous Publications Museum of Zoology, University of Michigan*, 132:1–182.

Vervust, B., Dongen, S.V., Grbac, I., and Van Damme, R. (2009). The mystery of the missing toes: extreme levels of natural mutilation in island lizard populations. *Functional Ecology,* 23:996–1003.

Vitt, L.J. (1983). Tail loss in lizards: the significance of foraging and predator escape modes. *Herpetologica*, 12:151–162.

Weatherhead, P.J., and Brown, G.P. (1996). Measurement versus estimation of condition in snakes. *Canadian Journal of Zoology,* 74:1617–1621.

Wikelski, M., and Thom, C. (2000). Marine iguanas shrink to survive El Niño. *Nature,* 403:37–38.

Wikelski, M., and Trillmich, F. (1997). Body size and sexual size dimorphism in marine iguanas fluctuate as a result of opposing natural and sexual selection: an island comparison. *Evolution,* 51:922–936.

Wingate, R.S. (1998). *A comparison of demography and morphological variation in two insular populations of the Bermuda rock lizard (Eumeces longirostris).* BSc Dissertation. University of Wales: Swansea, Wales.

## **APPENDIX 4A** – Bermuda skink age class determination (*Photo Credit: Turner, H. 2017*).

| Life Stage | Measurements | Description | Image |
|---|---|---|---|
| Hatchling | 50 -60 mm SVL<br>< 3.5 g | Hatchlings (< three months old) have bright blue tails and black and cream stripes extending from the head to the top of the tail with golden brown bodies (Wingate, 2005). They have short slender legs with five digits on each clawed foot. The skink's tail length is usually equal to or greater than, as much as 150 percent, of the animal's snout-vent length. |  |
| Juvenile | 48- 70 mm SVL<br>3.0 - 8.0 g | Juveniles (< one year old) progressively lose the blue colouration in their tails but still have bronzed dorsal coloration and two sets of dorsolateral black and cream stripes similar to hatchlings. |  |
| Sub-adult | 65- 85 mm SVL<br>7.5 - 21.0 g | Sub-adults (between one and two years old) no longer have any blue colouration on their tails. They look similar to juveniles with their dark brown to grey bodies and two distinct longitudinal black and yellow stripes on the dorsum, some also have orange chins and cheeks. |  |
| Adult | SVL > 72 mm - 95 mm<br>> 13.0 g – 30.0 g | Adults (> two years old) have a grey to black mottled dorsum with no striping, and salmon pink to vivid orange chin and cheeks. Venter colouration may vary between a beige, grey, yellow to orange colour. |  |

# CHAPTER 5

# MORPHOLOGICAL VARIATION BETWEEN ISOLATED BERMUDA SKINK *PLESTIODON LONGIROSTRIS* SUB-POPULATIONS

HELENA TURNER AND RICHARD A. GRIFFITHS

Adult skink on Nonsuch Island, Bermuda. (*Photo Credit: Dept. of Environment and Natural Resources, Bermuda, 2004*).

## 5.1 ABSTRACT

We examined intraspecific morphological variation between geographically isolated sub-populations of the endemic Bermuda skink (*Plestiodon longirostris*). We hypothesised that differences between the environments of isolated sub-populations (e.g. foraging opportunities and levels of predation) might create environmental or selective pressures resulting in morphological divergence between sub-populations. We present detailed morphometric data for seven sub-populations of *P. longirostris.* A total of ten linear biometric measurements were recorded from 1172 lizards. Morphometric variations were analysed using principal component and cluster analysis. Results indicated that seven sub-populations were clustered into three different groups: (1) Southampton Island, Nonsuch Island and Castle Island; (2) Charles Island, Spittal Pond (a mainland site) and Palm Island (located in the west of the Island chain); (3) Inner Pear Rock (located in the east of the island chain). Hind limb lengths and forearm lengths collected from skinks between 2015 and 2017 were also significantly shorter than those measured in earlier studies taken at least ten years before.

We demonstrate that morphological variation has occurred at the subpopulation level in response to different environmental pressures such as isolation, changes to habitat and predator levels over time, especially on these small offshore islands. Invasive species can prompt adaptive responses, altering the nature of interactions between invaders and the natives they contact.

## 5.2 INTRODUCTION

Phenotypic variation may reflect variation in fitness that arises through natural selection. Individual morphometric variation ultimately affects behaviour, growth rates, survivorship and reproductive success (Bjorklund, 1993; Post et al., 1997). These variables can be impacted as a result of isolation (e.g. Hendry et al., 2000), competition (e.g. Losos, 1994), predation (e.g. Langkilde, 2009), habitat change (e.g. Vitt et al., 1997), and environmental variables particularly temperature changes (e.g. Elphick and Shine, 1998).

Numerous approaches have been used to compare ecology and morphology among closely related species of lizards. For example, the study of limb size and proportions, which directly reflect changes in habitat and lifestyle may affect locomotor performance or fitness (Losos, 1990; Garland and Losos, 1994; Miles et al., 1995; Bonine and Garland, 1999). Therefore, lizard species may exhibit morphological specialisations that improve behavioural performance in the microhabitats they use (Melville and Swain, 2000). Given the widespread occurrence of these changes and their potential influence on the ways in which selection pressures translates into evolutionary shifts, we need a clearer understanding of the nature, magnitude and persistence of environmentally induced modifications. For example, on each of the Greater Antillean islands, different species of anoles occupying similar microhabitats tend to be similar in body size, limb and tail proportions, and other characteristics (Williams, 1972). Similar anole communities have evolved independently at least four times in the West Indies (Williams, 1983; Losos, 1992; Irschick et al., 1997; Losos et al., 1998), suggesting a strong and highly predictable relationship between lizard ecology (habitat use) and general morphology. However, studies of other groups of lizards do not confirm the patterns found in Caribbean anoles (e.g. Jaksic et al., 1980; Miles, 1994; Vanhooydonck and Van Damme, 1999). Although Caribbean *Anolis* lizards have been used in ecomorphological studies for many years they may not constitute a good model for other lizard groups (Vanhooydonck and Van Damme, 1999). Understanding differences in patterns of ecomorphological relationships is an important component for studying responses to the environment.

Lizards of the family Scincidae are mostly distributed over South-east Asia, the South Pacific and North America. They have radiated extensively into habitats ranging from heathland and mountains, to tropical forests, semi-desert and desert, but are absent from boreal and polar regions (Honda et al., 2000). Within these habitats, skinks occupy microhabitats that vary considerably. Some species are arboreal, frequently climbing shrubs or even trees (e.g. prehensile-tailed skinks *Corucia zebrata,* snake-eyed skinks *Cryptoblepharus virgatus*); some are saxicolous (e.g. sinbad skinks, *Oligosoma pikitanga*, five-lined mabuya *Mabuya quinquetaeniata*); some are cursorial and live on very open or highly vegetated terrain (e.g. open-litter rainbow skinks *Carlia pectoralis*, garden skinks *Lampropholis delicata*). However, many skinks are fossorial (e.g. Florida sand skinks *Neoseps reynoldsi,* Woodbush legless skink *Acontias rieppeli*), digging and burrowing in the ground.

Bermuda skinks (*Plestiodon longirostris*) are considered critically endangered throughout their range. Natural disasters such as tropical storms and hurricanes as well as anthropogenic activities resulting in habitat loss, plus the introduction of invasive flora and fauna have all contributed to small and isolated sub-populations across Bermuda (Raine, 1998). Given the variation in the availability and quality of resources especially among these small oceanic islands, with differing levels of shelter, prey availability, competition and increased predation it is thought morphological differences may occur amongst subpopulations (Davenport et al., 1997; Raine, 1998), since energy expenditure can be lowered by a reduction in body size (McNab, 1994). This suggests an important role of the environment in shaping variation in morphology. Isolation due to habitat loss exposes organisms to novel environmental challenges, often leading to reduced species richness and diversity due to small sub-populations (McNab, 1994). In this study, we examine the extent of morphological variation amongst seven isolated *P. longirostris* sub-populations in Bermuda.

## 5.3 METHODOLOGY

### *5.3.1 Lizard collection*

The Bermuda skink is a medium sized [adult snout to vent length (SVL): 64-95 mm and 13.0 – 30.0 g body mass] scincid lizard endemic to Bermuda (Glasspool and Outerbridge, 2004). Lizards were captured at various times between September 1997 and July 2017 (*See Supplementary Material 1*) across seven locations in Bermuda: Palm Island, Spittal Pond, Charles Island, Castle Island, Southampton Island, Nonsuch Island, and Inner Pear Rock (Fig. 5.1).

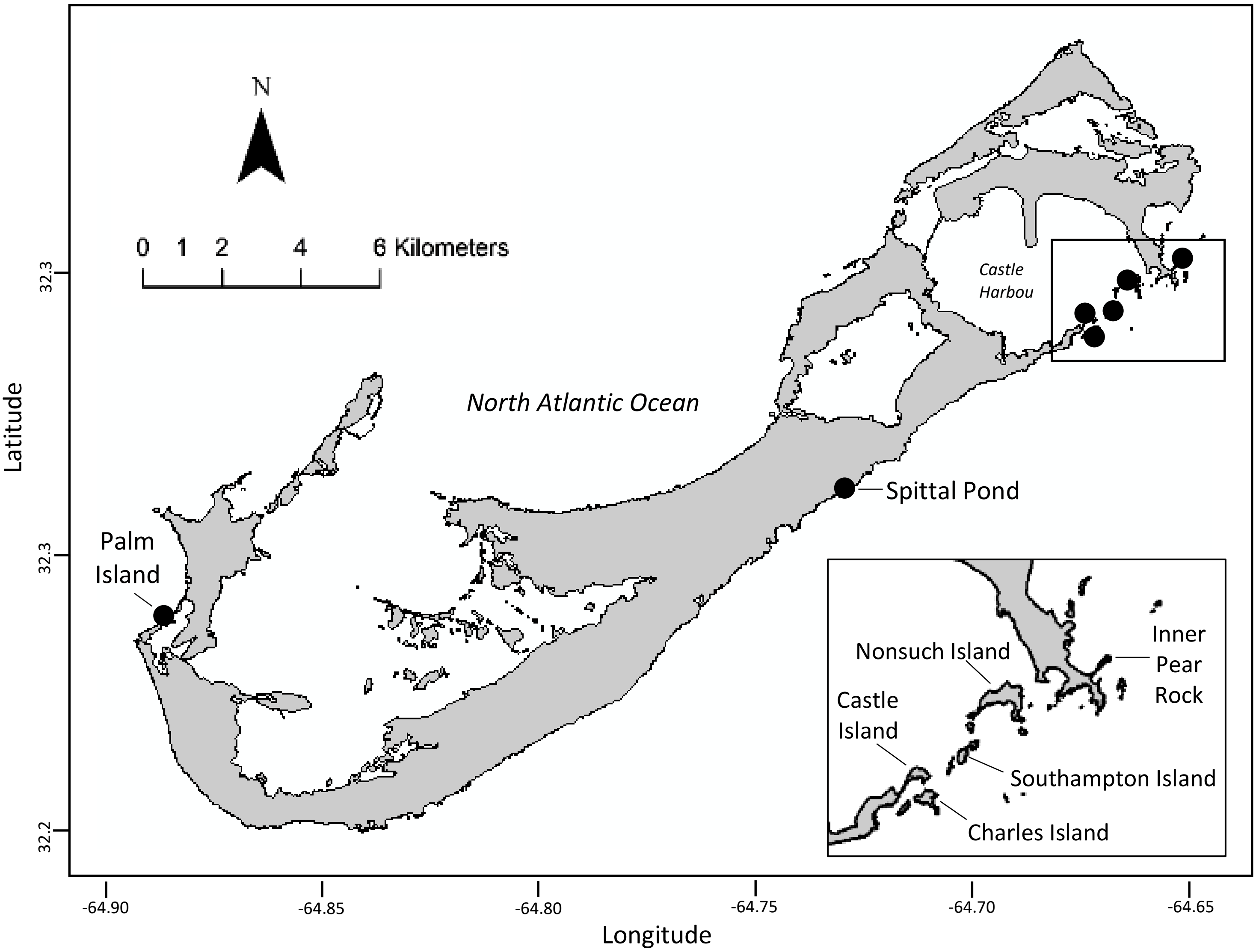


**FIGURE 5.1**. Bermuda skinks collected from seven locations 1998 – 2017 for morphometric analyses across Bermuda. (Images created in ArcMap 10.5).

The methodology was consistent between surveys and followed Davenport et al. (1997). Individuals were captured using glass jar pit fall traps (4.5 litres) with fish (Bumblebee® sardines in oil) as bait (Davenport et al., 1997; Glasspool and Outerbridge 2004). Traps were placed approximately every 5 – 10 metres, as this is considered to be within the skink's home range (Raine, 1998). Each trap was placed at a 45°angle against a rock to prevent escapes and covered with a large palmetto leaf or damp towel to prevent the traps overheating. Traps were checked hourly between 1100 and 1600 hrs (i.e. five checks per day). The number of traps per site (between 30 and 104) and the number of days of trapping (3-5 days) depended on the size of the site surveyed.

### *5.3.2 Lizard sampling and measuring*

The following nine linear biometric measurements were systematically recorded for each skink captured: head length (HL), head width (HW), forearm length (FL), fore digit length (FDL), hind limb length (HLL), hind digit length (HDL), tail width (TW), tail length (TL), and SVL (all described in further detail in Appendix 5A). Morphological measurements were recorded using digital callipers (precision: ± 0.01 mm), except for tail length and SVL which were measured with a ruler (to the nearest 1 mm). The ruler provided a flat surface to support the lizard, so they could be measured accurately, and also reduced the number of escapes. All measures of limbs and digits were taken from the right side of the skink unless there was mutilation in which case the left limb or digit was used. Body mass (BM) was also determined by placing each individual lizard in a polythene bag and weighing it using a 60 g Microline Pesola® spring balance (precision: ± 0.3 g).

Surveys were undertaken in two time periods, those in the 'Past' between 1997and 2004 by Davenport et al. (1997), Raine (1998), Wingate (1998), Hammond (2000) and Glasspool and Outerbridge (2004). Those surveys undertaken in the 'Present' were carried out by the author between 2015 and 2017.

Individuals captured in 'Present' surveys were identified by tagging with passive integrated transponder (PIT) tags (see Chapter 2). For individuals surveyed in the 'Past' studies, or those that were too small to be tagged (< 11.0 g), a spot of green coloured paint was applied to either the limbs or dorsal surface that would be detectable for the duration of the study, ensuring the same individuals were not recorded multiple times. Body scale colourations were recorded in order to divide specimens into three age classes: juveniles, sub-adults or adults, in accordance with existing knowledge of skink colouration (Raine 1998). As there is very little sexual dimorphism between Bermuda skinks this was not thought to impact morphological analyses. However, skinks that could be sexed by the presence or absence of the hemipenes, were recorded. To reduce the effects of age as a source of variation in the data, we only considered adult individuals in the morphometric analyses.

### *5.3.3 Statistical analyses*

A principal-components-based factor analysis (PCA) was used as different variables were measured on different scales (i.e. grams and millimetres), to reduce the dimensionality of the variation and to capture most of the variability in the data. SVL was used as the body length metric rather than total length (TOTL), to avoid errors resulting from many specimens having partly regenerated tails (Meiri, 2008). To assess the normality of the nine morphological variables (BM, SVL, HL, HW, FAL, FDL, HLL, HDL, TW) the Kolmogorov-Smirnov test was applied (Sokal and Rohlf, 1995). All variables were normally distributed ($P < 0.01$). Various axis rotations were then tested, and one was selected for the optimal separation of the sub-populations in multidimensional space (Kim and Mueller, 1978). A quartimax PCA was optimum due to simplifying the interpretation of the observed variables (Neuhaus and Wrigley, 1954). Ellipses were plotted with a 95% coverage. The principal components can be interpreted by examining which variables are most strongly correlated with each component, i.e. which of the eigenvalues is the farthest from zero in either direction. PCA computations were performed using R 3.3.2 using the following packages: doSNOW, ggpubr, readxl, dplyr, tidyr, SimComp, ggsci, factoextra, FactoMineR, lme4, ggsci, car, caret, lmtest, rcompanion, broom (R Core Team. 2016).

Morphometric variation was then analysed using a multivariate analysis of variance (MANOVA). This used all nine dependent variables, a type III sum of squares and used Wilks' lambda to compare multivariate sample means between the following groups (independent variables): between sex (males and females), time (past and present) and location (Palm Island, Spittal Pond, Charles Island, Castle Island, Southampton Island, Nonsuch Island, Inner Pear Rock). All morphological variables fulfilled the assumptions of normality (Shapiro-Wilk's test, $P > 0.05$ in all cases) and homogeneity of variances (Bartlett's test, $P > 0.005$ in all cases). MANOVA was followed by a post-hoc univariate ANOVA to assess which variables were significantly different between groups by comparing means (Zar, 1999). As only a limited number of individuals were sexed, a two-sample *t*-test was used to compare means between sexes. The level of significance was set at $P \leq 0{\cdot}05$ for all analyses, unless otherwise stated.

The relative dissimilarity of populations was assessed using cluster analysis and multi-dimensional scaling (MDS) with Primer v7 software (Clarke and Gorley, 2015). MDS analyses were considered complete when successive iterations decreased stress by less than 0.01. Cluster analyses were run using a normalised Bray-Curtis similarity index (with results given as percentile values) and MDS using normalised euclidean distances (Bray and Curtis, 1957).

## 5.4 RESULTS

### *5.4.1 Average body size and morphology*

In total, 1172 lizards were captured between 1997 and 2017 from seven sites across Bermuda (Southampton Island *n* = 638, Castle Island *n* = 249, Inner Pear Rock *n* = 41, Nonsuch Island *n* = 68, Charles Island *n* = 62, Palm Island *n* = 30, Spittal Pond *n* = 84). Of those caught, 66% were classed as adults (*n* = 773, 82.43 mm SVL, 19.42 g), 22% were classed as sub-adults (*n* = 261, 73.16 mm SVL, 13.65 g) and 12% were classed as juveniles (*n* = 138, 60.03 mm SVL, 8.13 g).

We found that using repeated measures (i.e. 50 skinks measured in the 'Present' by three different people), seven out of the nine morphological measurements yielded very high correlations between pairs of measures (Pearson's correlation coefficient: BM: $r = 0.955$, SVL: $r = 0.947$, HL: $r = 0.976$, HW: $r = 0.968$, FL: $r = 0.910$, HLL: $r = 0.906$, TW: $r = 0.934$, $n = 115$). [With hind digit and fore digit measurements the repeatability was slightly lower (HDL: $r = 0.697$, FDL: $r = 0.649$, $n = 115$) as error is often more significant for smaller measured values (Rabinovich, 2005)]. As 50% of lizards were found to have regenerated tails due to the varying degree of mutilations, tail length was removed from further analyses.

### *5.4.2 Principal component analysis*

The quartimax rotated PCA showed that over 80% of the variance in the data set was explained by the first three principal components (58.1% for PCA1, 17.9% for PCA2 and 4.3% for PCA3). Variables that load near 1 are clearly important in the interpretation of the factor, and variables that load near 0 are unimportant therefore we report any correlations with a value above 0.5 to simplify the interpretation of the factors (Bryant and Yarnold, 1995). We found that the first principal component strongly positively correlated with almost all morphometric variables: BM, SVL, HL, HW, FAL, FDL, HDL and TW apart from HLL (Table 5.1). This suggests that these eight criteria vary together i.e. if one variable increases, then the remaining variables tend to increase as well.

**TABLE 5.1.**

Eigenvalues of principal component axis with quartimax rotation of morphometric data. Correlations with a value above 0.5 are most strongly correlated and are shown in boldface.

| Variable | Principal component analysis with quartimax rotation | | |
|---|---|---|---|
| | PCA1. | PCA2. | PCA3. |
| BM | **0.868** | -0.169 | -0.074 |
| SVL | **0.506** | 0.102 | -0.126 |
| HL | **0.896** | -0.002 | 0.052 |
| HW | **0.882** | -0.058 | -0.050 |
| FAL | **0.589** | **0.516** | 0.285 |
| FDL | **0.562** | 0.425 | **-0.539** |
| HLL | 0.279 | **-0.756** | 0.446 |
| HDL | **0.588** | -0.076 | **0.507** |
| TW | **0.800** | 0.056 | -0.203 |
| Variance (%) | 58.1 % | 17.9 % | 4.3 % |

The second and third principal components each have two values with minor correlations. The second principal is correlated with forearm length (FAL) and hind limb length (HLL). As hind limb length (HLL) decreases, forearm length (FAL) increases and the remaining components show little effect. The third principal component is correlated with fore digit length (FDL) and hind digit length (HDL). As fore digit length decreases the hind digit length increases and the remaining components show little effect. This suggests that limb and digit lengths vary considerably between individuals. However, as PCA 3 only accounts for 4.3 % of explained variance this component is relatively unimportant.

Overall, the eigenvalues of principal components 1, 2 and 3, revealed some population-level morphological differentiation. We found variables were positively correlated and grouped together (whereas negatively correlated variables were positioned on opposite sides of the plot). There is a clear overlap between lizards from Nonsuch Island (NI), Southampton Island (SI) and Castle Island (CI), indicating close relationships. Lizards from Palm Island (PI), Spittal Pond (SP), Charles Island (CHI) and Inner Pear Rock (IP) are slightly more separated indicating some level of morphological differences, in particular between hind limb lengths and fore digit lengths (Fig. 5.2).

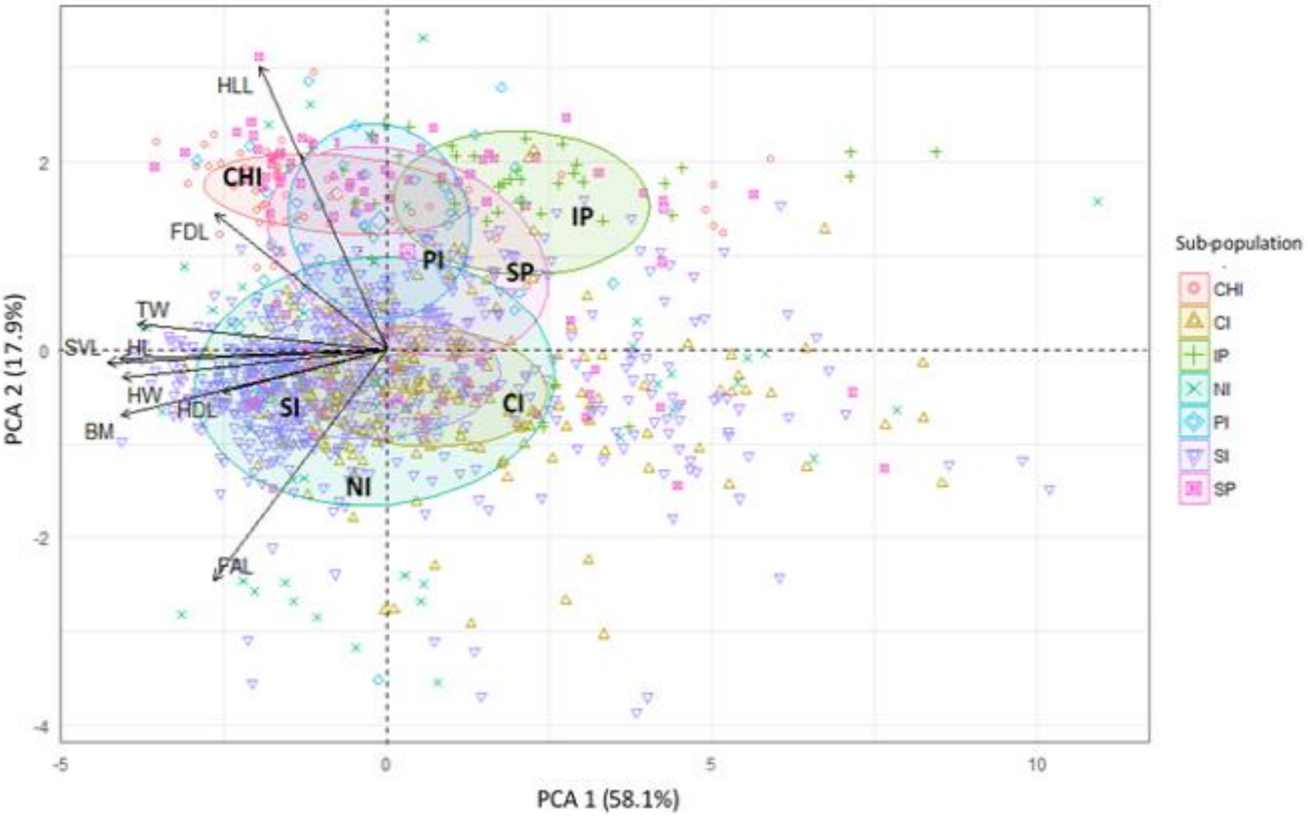


**FIGURE 5.2**. PCA1 vs. PCA2 of the principal component analysis showing differences between the seven locations (CHI = Charles Island, CI = Castle Island, IP = Inner Pear Rock, NI = Nonsuch Island, PI = Palm Island, SI = Southampton Island, SP = Spittal Pond). Each ellipse represents the region of graph space occupied by sub-population.

When using a PCA analyses to compare measurements over time, the results showed that time explains much variance. Individuals in the 'Present' exhibited significant positive correlations and therefore shorter hind limb lengths ($r = 0.709$, $P < 0.001$) and forearm lengths ($r = 0.531$, $P < 0.001$) than those in the 'Past' (Fig. 5.3).

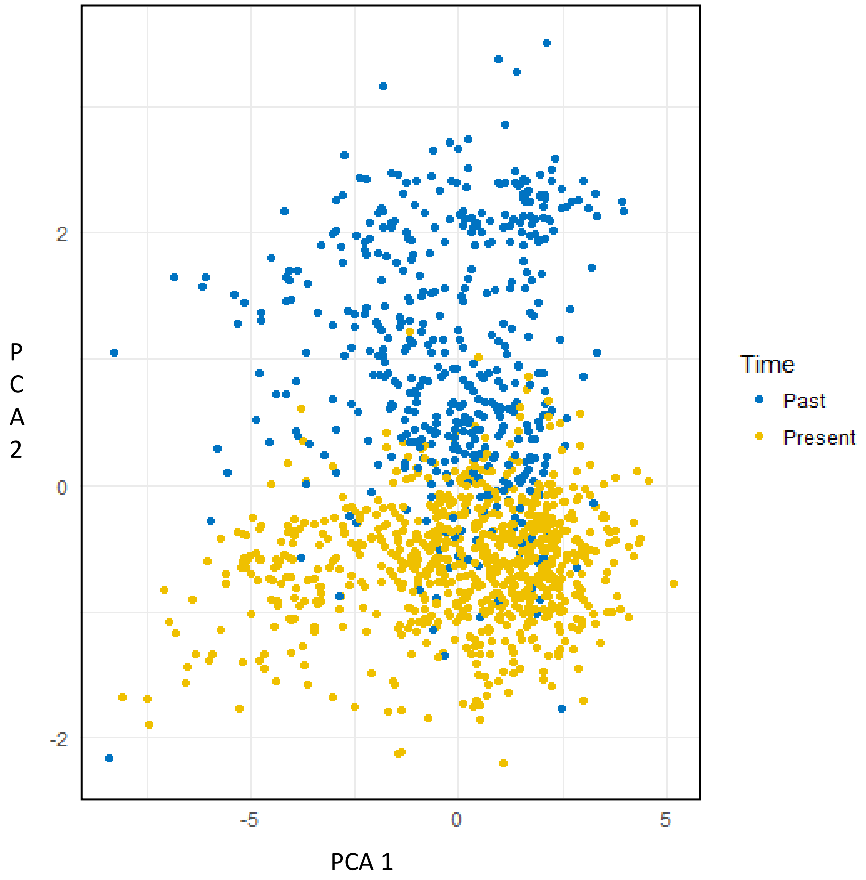


**FIGURE 5.3**. Association between time ('Past' and 'Present') and skink measurements where variation mostly accounts for differences between hind limb and fore digit lengths.

### *5.4.3 Multivariate analysis*

Using a MANOVA test, results show that independent variables were significantly associated with the linear combination of all dependent variables. This indicates that at least one of the measurements investigated was significantly different across time (F(9, 757) = 169.10, $P$ < 0.001) and site (F(54, 3864) = 33.91, $P$ < 0.001; Table 5.2).

There was no association between head length and hind digit length, which did not differ significantly between sites. Differences between measurements among other sites was mostly due to the smaller size of skinks on Inner Pear Rock. For example, skinks on Southampton Island had the greatest body mass (mean: 20.67 ± 0.53 g), whereas skinks on Inner Pear Rock had the lowest body mass (mean: 11.59 ± 0.41 g), SVL was greatest on Nonsuch Island (mean: 86.28 ± 0.33 mm) but lowest on Inner Pear Rock (mean: 73.61 ± 0.39 mm). Individuals on Inner Pear Rock had the lowest mean across all measures apart from fore digit length which Palm Island had the smallest mean (6.43 ± 0.15 mm and 5.76 ± 0.23 mm, respectively).

**TABLE 5.2.**
MANOVA test results comparing site and time between Bermuda skink morphological measurements.

| Measurement | Variable | Wilks' lambda (λ) | df | Err. df | *F* | *P* |
|---|---|---|---|---|---|---|
| BM | Site | 0.837 | 6 | 765 | 24.48 | **< 0.01** |
| | Time | 0.799 | 1 | 765 | 2.42 | **< 0.01** |
| SVL | Site | 0.859 | 6 | 765 | 20.65 | **< 0.01** |
| | Time | 0.861 | 1 | 765 | 10.11 | **< 0.01** |
| HL | Site | 0.764 | 6 | 765 | 22.16 | 0.413 |
| | Time | 0.852 | 1 | 765 | 17.42 | 0.309 |
| HW | Site | 0.896 | 6 | 765 | 14.46 | **< 0.01** |
| | Time | 0.671 | 1 | 765 | 20.53 | 0.092 |
| FAL | Site | 0.862 | 6 | 765 | 4.42 | **< 0.01** |
| | Time | 0.385 | 1 | 765 | 108.36 | **< 0.01** |
| FDL | Site | 0.891 | 6 | 765 | 15.38 | **< 0.01** |
| | Time | 0.783 | 1 | 765 | 14.68 | **< 0.01** |
| HDL | Site | 0.804 | 6 | 765 | 32.75 | 0.180 |
| | Time | 0.825 | 1 | 765 | 49.61 | **< 0.01** |
| HLL | Site | 0.396 | 6 | 765 | 233.80 | **< 0.01** |
| | Time | 0.597 | 1 | 765 | 11.89 | **< 0.01** |
| TW | Site | 0.722 | 6 | 765 | 10.76 | **< 0.01** |
| | Time | 0.785 | 1 | 765 | 3.39 | 0.087 |

All measures differed over time apart from head length, head width and tail width. All past measurements were significantly larger than in the present (BM = 19.95 g and 18.42 g, SVL = 82.97 mm and 80.49 mm, FAL = 17.60 mm and 12.57 mm, FDL = 6.79 mm and 6.37 mm, HDL = 12.48 mm and 11.84 mm, HLL = 25.65 mm and 22.70 mm, respectively). Skinks recorded from the 'Past' had an 8.31 % greater body mass and were 3.08 % longer SVL than skinks measured in the 'Present'.

### *5.4.4 Comparison between sexes*

Seventy-one percent of adults were sexed. In total, 252 individuals were female and 297 were male (Southampton Island *n* = 302, Castle Island *n* = 198, Nonsuch Island *n* = 30, Spittal Pond *n* = 19). The results show that there were significant differences in head width, head length, tail width, hind limb length and body mass between sexes. Females had a slightly larger body mass than males (19.74 g and 19.08 g, respectively) but males had larger SVLs (81.68 mm and 77.82 mm, respectively), head widths (13.60 mm and 12.55 mm, respectively), tail widths (9.82 mm and 9.20 mm, respectively), and hind limb lengths (22.41 mm and 20.96 mm, respectively; Table 5.3).

**TABLE 5.3.**
Two sample *t* -test results between adult male and female Bermuda skink and between nine morphological measurements. * Statistically significant at the $P < 0.05$ level after the Bonferroni adjustment (cut-off *P*-value = 0.005).

| | Male (*n* = 297) | Female (*n* = 252) | | |
|---|---|---|---|---|
| Measurement | Mean ± SEM | Mean ± SEM | *t* | *P* |
| BM | 18.08 ± 0.28 | 18.74 ± 0.31 | -4.00 | < 0.001* |
| SVL | 82.68 ± 0.43 | 77.82 ± 0.53 | -5.98 | < 0.001* |
| HL | 19.56 ± 0.13 | 18.33 ± 0.16 | -1.35 | 0.271 |
| HW | 13.60 ± 0.11 | 12.55 ± 0.12 | -6.44 | < 0.001* |
| FAL | 17.40 ± 0.13 | 16.27 ± 0.14 | 2.19 | 0.345 |
| FDL | 6.36 ± 0.051 | 6.13 ± 0.06 | -2.16 | 0.084 |
| HLL | 22.41 ± 0.15 | 20.96 ± 0.17 | -5.61 | < 0.001* |
| HDL | 12.58 ± 0.10 | 12.22 ± 0.09 | 0.65 | 0.063 |
| TW | 9.82 ± 0.071 | 9.20 ± 0.08 | -5.75 | < 0.005* |

### *5.4.5 Cluster analysis and multi-dimensional scaling*

The results showed three groupings, where Bermuda skink sub-populations differed morphologically (Fig. 5.4A and 5.4B). However, all sub-populations were found to be more than 96% similar with Inner Pear Rock individuals being the most dissimilar from all other sites.

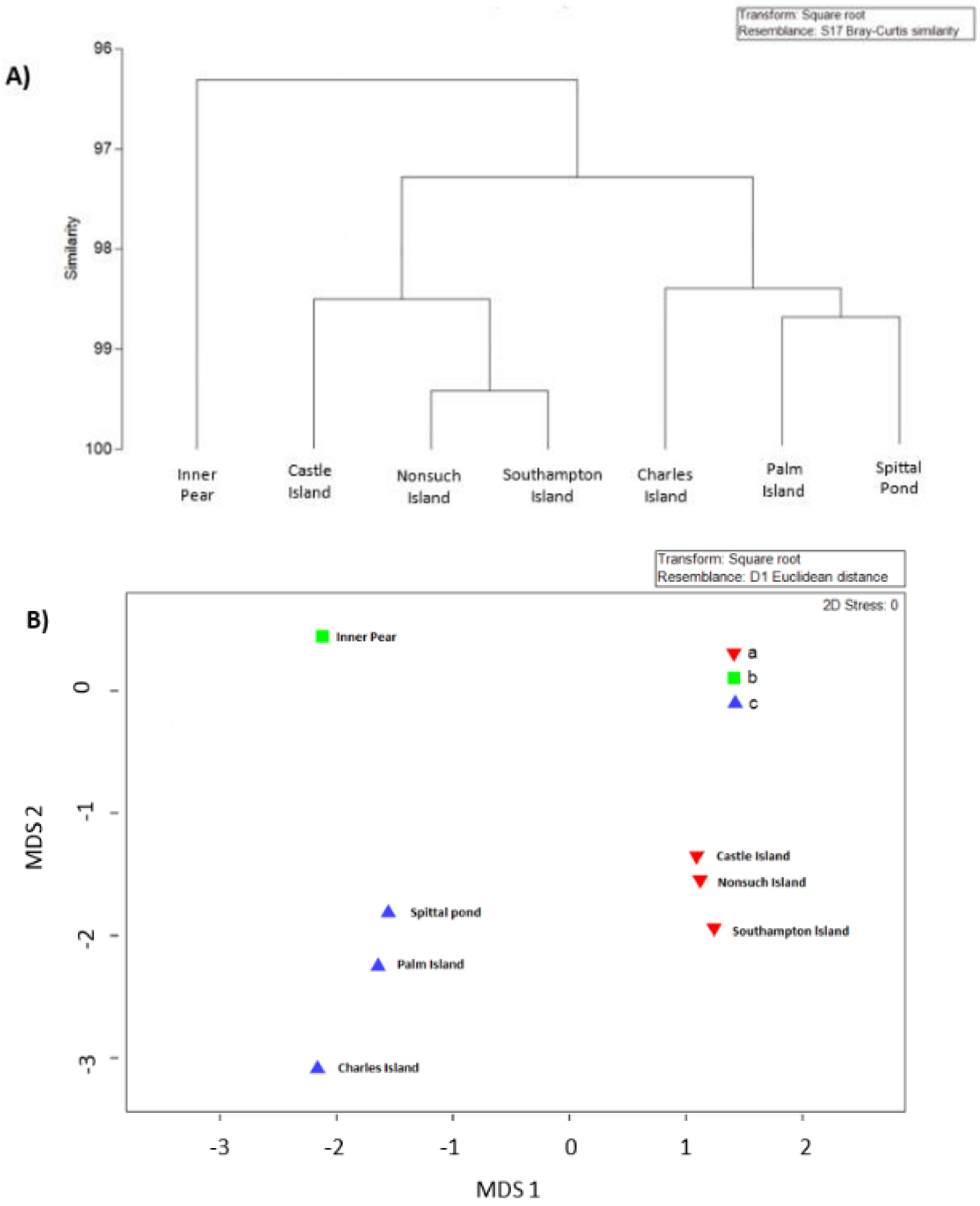


**FIGURE 5.4**. Using seven Bermuda skink continuous morphological features A) Cluster analysis B) Multidimensional scaling. (Euclidean distance, stress < 0.01).

## 5.5 DISCUSSION

### *5.5.1 Variation in lizard morphology*

We used multivariate analyses to compare morphologies between time and site, as well as to identify differences between sexes. The results revealed that there were differences between sub-populations. Although the majority of variation was attributed to general growth as the measurements increase together, skinks were found to have significantly shorter hind limb lengths and shorter forearm lengths in the 'Present' compared to the 'Past'.

Short hind limbs and forearms are often found in saxicolous (rock-dwelling) species (Snyder, 1962; Jaksic et al., 1980; Pounds, 1988; Sinervo and Losos, 1991). When skinks were abundant across Bermuda, they were found in variety of habitats including sand and dune, coastal rock and scrub and along cedar groves (Edgar et al., 2010). It is therefore highly likely that anthropogenic disturbances, increased predation and loss of habitat have pushed skinks into suboptimal habitat. This implies that selective pressures may be altering skink morphology, as shorter limbs on ground dwelling lizards will help increase endurance, agility and manoeuvrability, allowing them to move quicker between rocks or through dense and narrow vegetation (Zaaf and Van Damme, 2001; Herrel et al., 2002; Van Damme et al., 2003; Kolbe et al., 2012; Kaliontzopoulou, 2013). Therefore, limb morphology is likely to influence lizard escape behaviour (Schulte et al., 2004). Agile lizards can also catch insects more efficiently which may also be beneficial on these smaller islands with limited resources. Limb size and digit reduction has also been found to correlate with habitat utilisation such as substrate conditions and burrowing behaviour (e.g. in the sand-swimming lizard genus *Lerista* (Greer, 1990)). On the other hand, analyses of other lizard groups (*Liolaemus*: See Jaksic et al., 1980, Lacertidae: see Vanhooydonck and Van Damme, 2003) have failed to find clear correlations between limb morphology and habitat.

In a recent study, Irschick et al. (1997) showed that ecomorphological patterns found in Caribbean anoles differ dramatically from those in mainland habitats found across Central and South America. Those from the mainland population at Spittal Pond were not found to differ morphologically from island populations and, in comparison, were very similar to Palm island and Charles island individuals. However, they did differ in colour. Skinks on the mainland had very white venter colourations compared to those from island populations especially Inner Pear Rock which had bright orange venter colourations. As the colour differences are on the ventral surface, they are unlikely to be related to predator defence or attracting mates. However, they could be the result of carotenoid-based colours as a stress response, as observed in the common lizard (Fitze at al. 2009).

Skinks from Inner Pear Rock were also significantly smaller in almost all measurements (apart from fore digit length). However, this is most likely due to Inner Pear Rock being the smallest island (< 2000 m²), so its resources will be limited, which may limit skink growth (Ballinger and Congdon, 1980; McNab, 1994). The rate of evolution is often accelerated on smaller islands, and species can evolve to greater extremes in size, in this case shifting towards dwarfism rather than gigantism (Filin and Ziv, 2004). Additionally, it may be beneficial being a smaller size as individuals will be less conspicuous and escape predation more easily (Herczeg et al., 2009).

### *5.5.2 Sexual dimorphism*

In many reptile species, determining sex can be ambiguous as morphology can vary seasonally, behaviourally, or environmentally. Although identifying sex is difficult, it can often be determined by the presence/absence of hemipenes (Davis and Leavitt, 2007) horns or dewlaps, colouration differences or even morphological differences as males attain greater body and head sizes than females (Herrel et al., 2007). However, fewer than 10% of geckos, skinks, and chameleons exhibit sexual dimorphism (Cooper and Greenberg, 1992). There were, however, some subtle differences noticed in morphology between the sexes. Females had larger body mass than males, which is most likely due to surveys being undertaken during breeding season (May-June), when females are more likely to be carrying eggs (Shine 1989; Griffith, 1990). Males generally had larger SVLs, head widths, hind limb lengths and tail widths. Sexual dimorphism in head sizes is often attributed to male-male combat or it could be an adaptation for holding onto the female during copulation (Carothers, 1984; Vitt and Cooper, 1985). As the hemipenes are located at the base of the tail, males often have wider tail widths than females (Radder et al., 2001). Male lizards also frequently show relatively longer hind limbs and therefore greater speed than females (Castilla and Labra, 1998; Cooper Jr, 1999; Martin and Lopez, 2001; Cooper Jr and Vitt, 2000). Females on the other hand do not engage in territorial activities and generally avoid predation by being more secretive, so that selection on morphological traits will not occur as quickly as in males (Herrel et al., 2002). Understanding the differences between sexes will assist with managing the sex ratios in the current captive breeding programme.

### *5.5.3 Sub-population similarity*

Bermuda skinks grouped into three morphologically different sub-populations. Group A (Castle Island, Nonsuch island, and Southampton Island, located in Castle Harbour); Group B (Inner Pear Rock, located in the far East of the island); and Group C (Spittal Pond a mainland site and Palm Island and Charles Island). The three groups are geographically isolated from each other (minimum distance between Groups A and B = 0.88 km, between Groups B and C = 7.71 km and between Groups A and C = 5.48 km), so differences in morphology are presumably due to separation for long periods of time. However, each site within Group C is also isolated and it is possible that larger sample sizes from Palm Island and Charles Island sites reveal further morphological separation. Overall, Castle Island, Nonsuch Island and Southampton Island were the most similar to each other, whereas Inner Pear Rock was found to be the most distinct of the sub-populations, but there was still more than 96 % similarity amongst individuals. Undertaking genetic analyses in the future may help resolve these clusters in terms of evolutionary history.

The Bermuda skink is no longer widespread across Bermuda and now has a limited and patchy distribution, with 2,300 - 3,500 individuals estimated to be left in the wild (Edgar et al., 2010). We have found evidence to suggest that lizard morphology can vary in response to changing habitat characteristics, which will help to understand the effects that changes in habitat and predation may have on lizards. Progressive development of these traits implies that selective forces are operating and may be advantageous in certain environments.

## 5.6 LITERATURE CITED

Ballinger, R.E., and Congdon, J.D. (1980). Food resource limitation of body growth rates in *Sceloporus scalaris* (Sauria: Iguanidae). *Copeia,* 1980:921–923.

Bjorklund, M. (1993). Phenotypic variation of growth trajectories in finches. *Evolution*, 47:1506–1514.

Bonine, K.E., and Garland, T. Jr. (1999). Sprint performance of phrynosomatid lizards, measured on a high-speed treadmill, correlates with hind limb length. *Journal of Zoology*, 248–255–265.

Bray, J.R., and Curtis, J.T. (1957). An ordination of the upland forest communities of Southern Wisconsin. *Ecological Monographs*, 27:325–349.

Bryant, F.B., and Yarnold, P.R. (1995). Principal-components analysis and confirmatory factor analysis. In: Grimm LG., and Yarnold PR (Eds.) *Reading and understanding multivariate statistics*. Pp. 99–136. American Psychological Association: Washington, DC, United States.

Carothers, J.H. (1984). Sexual selection and sexual dimorphism in some herbivorous lizards. *American Naturalist*, 124:244–254.

Castilla, A.M., and Labra, A. (1998). Predation and spatial distribution of the lizard *Podarcis hispanica* atrata: an experimental approach. *Acta Oecologica*, 19:107–114.

Clarke, K.R., and Gorley, R.N. (2015). *PRIMER v7: User Manual/Tutorial*. PRIMER-E: Plymouth, United Kingdom.

Cooper, W.E Jr. (1999). Trade-offs between courtship, fighting and anti-predatory behaviour by a lizard, *Eumeces laticeps*. *Behavioral Ecology and Sociobiology*, 47:54–59.

Cooper, W.E Jr., and Greenberg, N. (1992). Reptilian colouration and behaviour. In: Gans C., and Crews D (Eds.) Pp 298–422. *Hormones, brain, and behavior (biology of the reptilia, vol 18)*. University of Chicago Press: Chicago, United States.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). *A study of populations of the Bermudian rock lizard (skink), Eumeces longirostris, cope* (1861), *on the islands of Nonsuch and Southampton, Bermuda*. Department of Agriculture and Fisheries: Bermuda.

Edgar, P., Kitson, L., Glasspool, A. and Sarkis, S. (2010). *Recovery plan for the Bermuda skink Eumeces longirostris*. Government of Bermuda, Department of Conservation Services: Bermuda.

Elphick, M.J., and Shine, R. (1998). Longterm effects of incubation temperatures on the morphology and locomotor performance of hatchling lizards (*Bassiana duperreyi*, Scincidae). *Biological Journal of the Linnean Society*, 63:429–447.

Filin, I., and Ziv, Y. (2004). New theory of insular evolution: unifying the loss of dispersibility and body-mass change. *Evolutionary Ecology Research*. 6:115–124.

Fitze, P.S., Cote, J., San-Jose, L.M., Meylan, S., Isaksson, C., Andersson, S., Rossi, J., and Clobert, J. (2009). Carotenoid-based colours reflect the stress response in the common lizard. *PloS ONE*, 4: e5111.

Garland, T. Jr., and Losos, J.B (1994). Ecological morphology of locomotor performance in squamate reptiles. In: Wainwright PC., and Reilly SM (Eds.) *Ecological morphology: integrative organismal biology*: Pp 367. University of Chicago Press: Chicago, United States.

Greer, A.E. (1990). Limb reduction in the scincid lizard genus *Lerista*. 2. Variation in the bone complements of the front and rear limbs and the number of postsacral vertebrae. *Journal of Herpetology*, 24:142–150.

Griffith, H. (1990). Miniaturization and elongation in *Eumeces* (Sauria: Scincidae). *Copeia*, 751–758.

Glasspool, A., and Outerbridge, M. (2004). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island*, Castle Harbour. Bermuda Zoological Society: Bermuda.

Hammond, M.P. (2000). *An analysis of Bermuda Skink (Eucemes longirostris) populations on Castle Island, Bermuda.* Bermuda Aquarium, Museum and Zoo: Bermuda.

Herrel, A., Meyers, J.J., and Vanhooydonck, B. (2002). Relations between microhabitat use and limb shape in phrynosomatid lizards. *Biological Journal of the Linnean Society*, 77:149–163.

Herrel, A., McBrayer, L.D., and Larson P.M. (2007). Functional basis for sexual differences in bite force in the lizard *Anolis carolinensis. Biological Journal of the Linnean Society*, 91:111–119.

Hendry, A. P., Wenburg, J.K., Bentzen, P., Volk, E.C., and Quinn, T.P. (2000). Rapid evolution of reproductive isolation in the wild: Evidence from introduced salmon. *Science,* 290:516.

Herczeg, G.B., Gonda, A.L. Merilä, J. (2009). Evolution of gigantism in nine-spined sticklebacks. *Evolution*, 63:3190–3200.

Honda, M., Hidetoshi, O., Kobayashi, M., Nabhitabhata, J., Yong, H.S., and Hikida, T. (2000). Phylogenetic relationships, character evolution, and biogeography of the subfamily lygosominae (Reptilia: Scincidae) inferred from mitochondrial DNA sequences. *Molecular Phylogenetics and Evolution*, 15:452– 461.

Irschick, D.J., Vitt, L.J., Zani, P., and Losos, J.B. (1997). A comparison of evolutionary radiations in mainland and Caribbean *Anolis* lizards. *Ecology*, 78:2191–2203.

Jaksic, J.F.M., Núnez, H., and Ojeda. F.P. (1980). Body proportions, microhabitat selection, and adaptive radiation of Liolaemus lizards in central Chile. *Oecologia*, 45:178–181.

Kaliontzopoulou, A., Bandeira, V., and Carretero, M.A. (2013). Morphological and locomotor sexual dimorphism in podarcis. *Journal of Zoology,* 289:294–302.

Kim, J. O., and Mueller, C. W. (1978). Introduction to factor analysis: What it is and how to do it. *Sage,* Beverly Hills, California, United States.

Kolbe, J.J., Leal, M., Schoener, T.W., Spiller, D.A., and Losos, J.B. (2012). Founder effects persist despite adaptive differentiation: A field experiment with lizards. *Science*, United States.

Losos, J.B. (1990). Ecomorphology, performance capability, and scaling of West Indian *Anolis* lizards: an evolutionary analysis. *Ecological Monographs*, 60:369 –388.

Losos, J.B. (1992). A critical comparison of the taxon-cycle and character-displacement models for size evolution of *Anolis* lizards in the Lesser Antilles. *Copeia*, 279–288.

Losos, J.B. (1994). Integrative approaches to evolutionary ecology: *Anolis* lizards as model systems. *The Annual Review of Ecology, Evolution, and Systematics*, 25:467–493.

Losos, J.B., Jackman, T.R., Larson, A., de Queiroz, K., and Rodríguez-Schettino, L. (1998). Contingency and determinism in replicated adaptive radiations of island lizards. *Science*, 279:2115–2118.

McNab, B.K. (1994). Resource use and the survival of land and freshwater vertebrates on oceanic islands. *The American Naturalist*, 144:643–660.

Meiri, S. (2008). Evolution and ecology of lizard body sizes. *Global Ecology and Biogeography*, 17:724–734.

Melville, J., and Swain, R. (2000). Evolutionary relationships between morphology, performance and habitat openness in the lizard genus *Niveoscincus* (Scincidae: Lygosominae). *The Biological Journal of the Linnean Society*, 70:667– 683.

Miles, D.B. (1994) Covariation between morphology and locomotory performance in Sceloporine lizards. In: Vitt LJ., and Pianka ER (Eds.), *Lizard ecology: historical and experimental perspectives*. Pp 207–235. Princeton University Press: Princeton, New Jersey, United States.

Miles, D.B., Fitzgerald, L.A., and Snell, H.L. (1995). Morphological correlates of locomotor performance in hatchling *Amblyrhynchus cristatus*. *Oecologia,* 103:261–264.

Neuhaus, J.O., and Wrigley, C. (1954). The quartimax method. *British Journal of Statistical Psychology*, 7:81–91.

Post, E., Stenseth, N.C., Langvatn, R., and Fromentin, J.M. (1997). Global climate change and phenotypic variation among red deer cohorts. *Proceedings of the Royal Society of London* B, 264:1317–1324.

Pounds, J.A. (1988). Ecomorphology, locomotion, and microhabitat structure: patterns in a tropical mainland Anolis community. *Ecological Monographs*, 58:299–320.

R Core Team. (2016). *R: A language and environment for statistical computing.* The R Foundation for Statistical Computing. Vienna, Austria.

Rabinovich, S.G. (2005). *Measurement errors and uncertainties: theory and practice*. Springer-Verlag: New York, United States.

Radder, R.S., Shanbhag, B.A., and Saidapur, S.K. (2001). Ontogeny of sexual size dimorphism in the tropical garden lizard, *Calotes versicolor* (Daud.). *Journal of Herpetology*, 35:156–160.

Raine, A. (1998). *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the Critically Endangered Bermuda Rock Lizard* (*Eumeces longirostris*). MSc Dissertation. University College London: United Kingdom.

Schulte, J.A., Losos, J.B., Cruz, F.B., and Nunez, H. (2004). The relationship between morphology, escape behaviour and microhabitat occupation in the lizard clade Liolaemus (Iguanidae: Tropidurinae: Liolaemini). *The Journal of Evolutionary Biology*, 17:408–420.

Shine, R. (1989). Ecological causes for the evolution of sexual dimorphism: a review of the evidence. *Quarterly Review of Biology*, 64:419–461.

Sinervo, B., and Losos, J.B. (1991). Walking the tight rope: arboreal sprint performance among *Sceloporus occidentalis* lizard populations. *Ecology*, 72:1225–1233.

Sokal, R.R., and Rohlf, F.J. (1995). *Biometry: the principles and practice of statistics in biological research.* Third Edition. W.H. Freeman: San Francisco, United States.

Snyder, R.C. (1962). Adaptations for bipedal locomotion of lizards. *American Zoologist*, 2:191–203.

Van damme, R., Vanhooydonck, B., Aerts P., and De Vree, F (2003) Evolution of lizard locomotion: context and constraint. In: Bels V., Gasc JP., and Casinos A. (Eds.) Pp 267-282. *Vertebrate biomechanics and evolution*. BIOS Scientific Publishers: Oxford, United Kingdom.

Vanhooydonck, B., and Van Damme, R. (1999). Evolutionary relationships between body shape and habitat use in lacertid lizards. *Evolutionary Ecology Research*, 1:785–805.

Vanhooydonck, B., and Van Damme, R. (2003). Relationships between locomotor performance, microhabitat use and antipredator behaviour in lacertid lizards. *Functional Ecology*, 17:160–169.

Vitt, L.J., Caldwell, J.P., Zani, P.A., and Titus, T.A. (1997). The role of habitat shift in the evolution of lizard morphology: evidence from tropical *Tropidurus*. *PNAS,* 94:3828–3832.

Vitt, L., and Cooper, W.E. (1985). The evolution of sexual dimorphism in the skink *Eumeces laticeps*: an example of sexual selection. *Canadian Journal of Zoology*, 63:995–1002.

Williams, E.E. (1972). The origin of faunas: evolution of lizard congeners in a complex island fauna. A trial analysis. *Evolutionary Biology,* 6:47–89.

Williams, E.E. (1983). Ecomorphs, faunas, island size, and diverse end points in island radiations of Anolis. In: Huey RB., Pianka ER., and Schoener TW. (Eds.) *Lizards ecology: studies of a model organism*. Pp. 326–370. Harvard University Press: Cambridge, Massachusetts, United States.

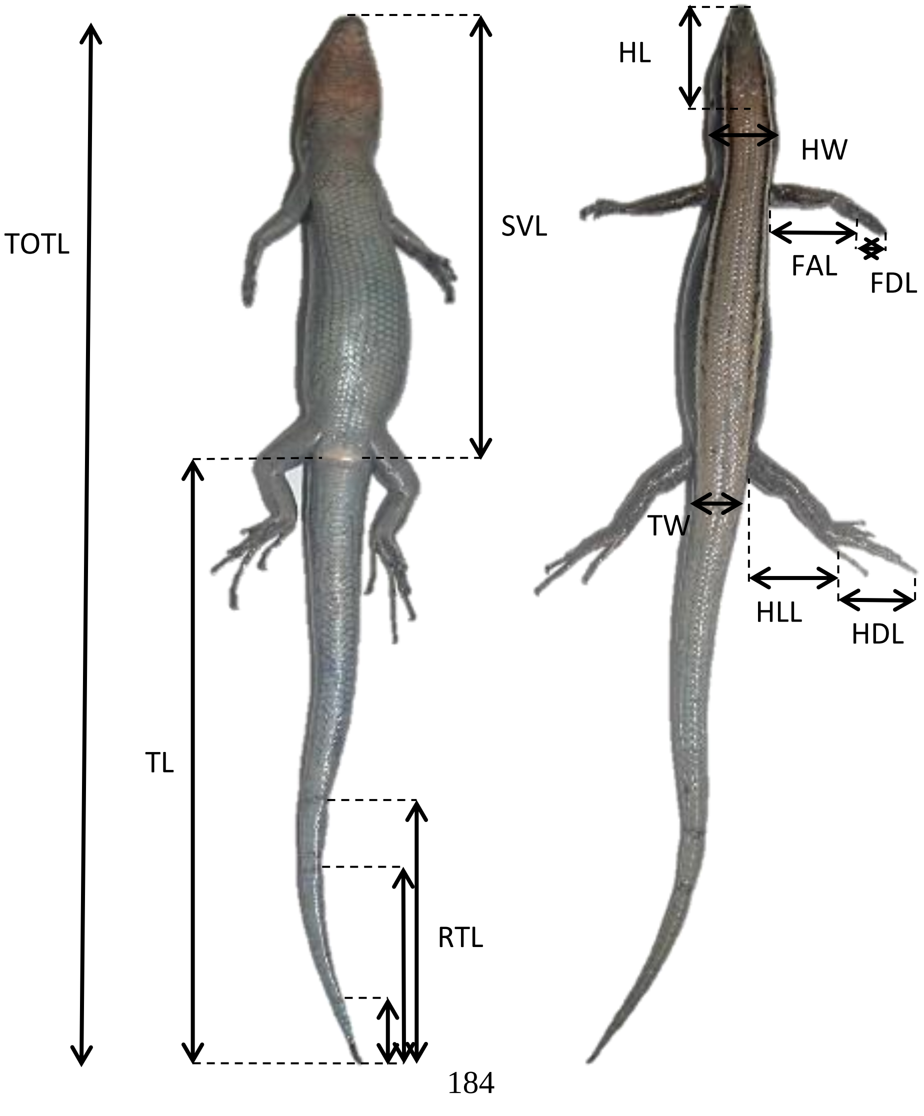
Wingate, R. (1998). *A comparison of demography and morphological variation in two insular populations of the Bermuda rock lizard* (*Eumeces longirostris*). BSc Dissertation. University of Swansea: Wales.

Zaaf, A., and Van Damme, R. (2001). Limb proportions in climbing and ground-dwelling geckos (*Lepidosauria*, Gekkonidae): a phylogenetically informed analysis. *Zoomorphology*, 121:45–53.

Zar, J.H. (1999). *Biostatistical analysis*. Fourth Edition. Prentice Hall, Upper Saddle River, New Jersey, United States.

## **APPENDIX 5A -** Description of the ten linear morphometric measurements taken on Bermuda skinks.

- *Snout to vent length* (SVL) was measured from the tip of the snout to the posterior end of the cloaca.
- *Head length* (HL) was measured from the tip of the snout to the posterior edge of the tympanum.
- *Head width* (HW) was measured at the widest part of the hinge of the jaw.
- *Tail width* (TW) was measured at its widest point just below the cloaca.
- *Tail length* (TL) was measured from the anterior end of the cloaca to the tip of the tail.
- *Fore limb length* (FLL) and *hind limb length* (HLL) were measured as the distance from the proximal end to the start of the fourth digit.
- *Fore digit length* (FDL) and *hind digit length* (HDL) were measured as the distance from the point where the fourth digit meets the palm to the extremity of the fourth digit, excluding the claw.
- *Regenerated tail length* (RTL) was from the point of blastema to the tail tip.
- *Total length* (TOTL) was measured from the tip of the snout to the tip of the tail.



# CHAPTER 6

# GENERAL DISCUSSION

Hatchling Bermuda skink. (*Photo credit: Chester Zoo, 2017*).

## 6.1 GENERAL DISCUSSION

### *6.1.1 Summary of the study*

Presence-absence surveys and mark-capture-recapture surveys were undertaken across Bermuda in 2015 – 2017 to investigate the current population status of the critically endangered Bermuda skink. PIT tags were used for the first time with this species to initiate long term monitoring. As a result, skinks were detected at 13 sites including two new sites, ranging from 1 - 692 captures per site. Two sites, Castle Island and Southampton Island located in Castle Harbour, are the main strongholds. However, the populations fluctuate in size, possibly in response to environmental conditions. All skink populations are affected by the effects of habitat loss, increased predation and anthropogenic threats, which have resulted in a decline in body condition and variations in morphology over time. Compared to surveys carried out at least 14 years earlier, skinks have become more restricted in their distribution, with the main strongholds being small uninhabited islands. However, even these stronghold populations are threatened by invasive species and extreme weather events.

Surveys for skinks were made at 40 sites across Bermuda, but they were only observed at 20% of these sites. However, predicted occupancy was estimated at about 25%. More intensive surveys may therefore be needed to detect all remaining sub-populations, so they can be managed appropriately. A minimum of five surveys is needed for a 95% probability of skinks being detected at a site. Skinks were more likely to be detected on undisturbed rocky coastal offshore islands that featured the presence of seabirds and prickly pears. In contrast, skinks were unlikely to be detected on sites with cat and rat predators (*Chapter 3*). Covariates revealed potential drivers of population decline, as well as indicating key priority areas for the conservation of the Bermuda skinks. The focus on restoring native coastal habitat on offshore islands and removal of invasive flora and fauna are priorities for the future. Testing the success of different management techniques to eradicate invasive species on isolated islands would be beneficial.

Capture-mark-recapture (CMR) surveys and robust design modelling found the two largest populations (on Castle Island and Southampton Island) fluctuated in size over the three-year period, and were threatened by increasing anthropogenic activities, invasive species, and habitat loss (*Chapter 2*). Undertaking intensive CMR surveys at all other known populations across Bermuda would help to identify the main threats at each site.

We demonstrate that differences between the isolated sub-populations (e.g. foraging opportunities and levels of predation) may create environmental or selective pressures resulting in morphological divergence, especially on the small offshore islands. As a result, individuals had smaller hind limb and forearm lengths than they did 1-2 decades previously. Out of seven sub-populations, three distinct groups were apparent (*Chapter 4*). Further genetic work using microsatellite analyses or next generation sequencing will determine the extent of gene flow and genetic variation between sub-populations, which will be vital to assist with population management in the future.

Additionally, human-induced habitat deterioration may negatively affect the body condition of Bermuda skinks over time, possibly as a result of skinks being forced to use unsuitable habitat. Indeed, skinks in suboptimal forested habitats had lower body condition than those from more favourable coastal rock and scrub habitats (*Chapter 5*). Measuring body condition can provide a surrogate measure of an animal’s fitness, therefore, the continuation of the collection of biometric data would be valuable to help to help build up a bigger picture to compare populations dynamics across other sites and over a longer period of time.

### *6.1.2 Limitations of this study*

PIT tagging was a satisfactory marking tool for the long-term monitoring of Bermuda skinks. However, only individuals over 11 g were tagged during this study to minimise the risk of reduced survival and tag loss (Welch et al., 2007; Smircich and Kelly, 2014). Current literature mentions as a rule of thumb that PIT tags should be no more than 2% of an animal’s body mass (Jepson et al., 2005; Smircich and Kelly, 2014). However, this is rule is largely based on fish studies. Currently the American Society of Ichthyologists and Herpetologists suggests an upper size limit of 10% of body mass for implanted devices in amphibians and reptiles but add that this percentage is clearly inappropriate for small species (ASIH, 2004). Throughout the study of Bermuda skinks, tags did not exceed 0.25% of the animal’s body mass so were well within current guidelines and did not appear to have any detrimental effects. Further studies should confirm the long-term effects of PIT tagging small skink species particularly regarding tag retention and longevity.

Juveniles are an important life stage to monitor in order to estimate critical population parameters, such as population viability. However, because individuals are small, secretive, and rarely caught, therefore data on juveniles is difficult to obtain (Pike et al, 2008). In this study, capture rates of juveniles were low and were deemed too small to be PIT tagged with the current 8 mm x 1.4 mm tags. Consequently, they were excluded from all analyses comparing variations over time. Although it would be important to account for juveniles to reduce any bias in future studies (which could be achieved by using a combination of smaller PIT tags and capturing hatchling individuals from artificially created nests) a study estimating juvenile reptile survival rates found they were highly correlated with adult survival rates (Pike et al, 2008).

A major issue encountered in the study concerned logistics. Accessing and landing on offshore islands was especially difficult since most are surrounded by rock pinnacles and subject to significant sea swell. Therefore, settled marine conditions are required which can be difficult to predict. Consequently, the number of survey days available is much reduced and regular surveying undermined. New survey techniques, such as the use drones are becoming more popular methods for monitoring isolated and difficult to access populations (Koh and Wich, 2012) as well as the use of camera traps for monitoring the behaviour of elusive and secretive species (Rowcliffe et al., 2008; Burton et al., 2015). With further development, these techniques could potentially be implemented more easily, and in some cases less expensively (i.e. with reduced human resources and boat costs) than the current methods. Additionally, as current trapping techniques include the use of bait, sampling relies on skinks actively foraging for food. Camera traps may be better able to monitor natural behaviours during other activity periods. Equally, surveying isolated locations or those that are difficult or unsafe to land on may be easier using drones.

### *6.1.3 Implications for populations*

In the early 1900s the dune formations across Bermuda were analysed. The resulting layers of sediment indicated that during the Pleistocene era (when the skinks arrived at Bermuda) Bermuda would have been about 518 km$^2$ compared to the present-day area of 55 km$^2$. Consequently, Bermuda has significantly decreased in size due to the melting of the Pleistocene ice caps thereby increasing sea levels (Olsen et al., 2006). The dunes showed that conditions in Bermuda during this time would have been stormier and even colder than the present climate as shown by a distinct lack of native vegetation present, which would not only aid the dune formation but equally explains why soil formation was restricted in Bermuda (Verrill, 1902; Sayles, 1931). Modern sand dunes are rare (Sayles, 1931), so Bermuda's pre-colonial habitat would have been predominately made of dunes and rocky coastal habitat with some native vegetation which has shown throughout this study as being the most suitable habitat for the Bermuda skinks (*Chapters 2, 3, 4* and *5*). Dense forest habitat appeared to have a negative impact, attracting predators such as rats that prefer to forage in dense understorey vegetation and adapt their feeding habitats to the available food types (Williams et al., 2003). Crows are also highly adaptable especially in their habitat requirements, and although their nesting sites are shifting from rural to urban areas, they still prefer to nest in densely forested areas occupying both deciduous and evergreen trees (Gorenzel and Salmon, 1992), which would not provide suitable basking or foraging spots for the skinks. Based on the results of this study, there are many threats that have played a significant role in the decline of the Bermuda skink and this study has provided a strong basis for progressing Bermuda skink conservation. For example, the removal of invasive flora and fauna and restoration of native habitat is highly recommended. Bermuda has a large human population with concomitant development of housing, roads and other infrastructure, thus reducing the size of available habitat as well as causing increased anthropogenic disturbances. To help mitigate these threats, artificial burrows, stone walls, rockeries or refugia should be created to increase the number of safe places for skinks to bask, escape predation and nest. Such artificial structures could prove to be important resources for skinks especially during cyclic weather events such as El Niño.

Additionally, conservation activities involving seabird colonies (white-tailed tropicbirds and Bermuda petrels) may help to increase food abundance (e.g. in the form of carrion or failed bird eggs) for Bermuda skinks that co-exist with them (Edgar et al., 2010). An abundance of fossil snail shells can also be found in the deep soil layers (Sayles, 1931), indicating a plentiful supply during pre-colonial times. This suggests that recovery of native snail populations should also be promoted.

Vital skink habitat and nesting areas can be impacted due to anthropogenic disturbances especially the construction of 'rock cairns' (rock stacks balanced on top of each other) or 'tombstoning', jumping into the sea from the coastal edges). As a result, access to the protected nature reserves within Castle Harbour should be restricted 'by license only' or via supervised tour groups to protect these fragile islands. Restricting access also allows bio secure methods to be implemented to prevent further invasive species being introduced, which could ultimately cause further decline to Bermuda's species by destroying natural habitats, outcompeting, depredating, or poisoning native taxa (Mooney and Cleland, 2001). For example, fire ants (*Solenopsis* sp.) were accidently introduced to the US in the 1930s and their spread has impacted agriculture and public health (Langkilde, 2009). Additionally, they have they been reported to prey on fence lizards by lifting up a scale using their mandibles and envenomating them by inserting their sting shaft into the underlying soft skin. If a lizard fails to respond to such an attack it will become paralyzed and die (Langkilde, 2009). As both eggs (Moulis, 1997) and young (Allen et al., 1997) of reptiles are often targeted by fire ants, and they have been identified as a primary cause of extirpation of the Texas horned lizard (*Phrynosoma cornutum*) (Goin, 1992). In some cases, lizards have been shown to rapidly adapt to invaders in order to survive in the face of a novel predator (Langkilde, 2009). However, due to low genetic diversity present in the Bermuda skinks (Brandley et al., 2010) the population will likely struggle to adapt to these changes (Templeton et al., 2001).

Although the Earth's climate will continue to change, climatic changes in the distant past were driven by natural causes, such as variations in the Earth's orbit or the carbon dioxide ($CO_2$) content of the atmosphere. Now climatic variation is undergoing a period of rapid changes, enhanced due to anthropogenic activities during the past 100 years. Many activities associated with human economic development have changed our physical and chemical environment in ways that modify natural resources. Seemingly small, human-induced changes to the natural greenhouse gases are typically projected to result in a global warming of about 1.5°C to 6°C in the 21st century (Blaustein et al., 2010). This range, especially if beyond 2–3°C, would likely result in ecologically significant changes (Fischlin et al., 2007). Additionally, the oceans uptake of atmospheric $CO_2$ will alter pH levels making it difficult for marine life to adapt (Hoegh-Guldberg et al., 2007; Doney et al., 2016). Global climate change is already having significant effects on species and ecosystems.

Climate change is therefore likely to impact Bermuda through changes in patterns of precipitation, levels of ultraviolet radiation, increases in tropical storms and a rising sea level. In turn, these will result in coastal erosion and loss of suitable habitat. Management options currently available include protecting coastal wetlands to allow for sea level rise; reducing the risks to wildlife from potential catastrophic events; and taking climate change into consideration when selecting the location and other characteristics of conservation areas. Ignoring climate change is likely to result in failure to reach wildlife management objectives. Assessing the vulnerability of populations to climate change and life on low-lying islands will inform decisions concerning whether skinks in such habitats (i.e. North and South Cock rocks) should be translocated to safer sites.

### *6.1.4 Future research*

This study is the first to undertake multi-year monitoring using CMR and PIT tagging methods at multiple sites across Bermuda. Future studies should continue long-term monitoring to identify whether the observed trends or fluctuations in population dynamics (see *Chapter 2 or 3*), or changes in body condition (see *Chapter 4*), or morphological features (see *Chapter 5*) are population specific depending on the threats they face.

Trained ’detection’ dogs have been used to determine the presence of endangered species and invasive species (Wasser et al., 2012). Detection dogs can be used within citizen science programmes (with the use of smartphone apps) to record sightings (Bonney et al., 2014). Using radio-tracking tags could also be a useful tool to determine the skink movement patterns and microhabitat use. This would create a more detailed understanding of the habitat and environmental changes that affect skink populations, which could inform conservation efforts and management. Radio transmitters have been attached to lizards in a variety of different ways including surgical implants, direct attachment to the dorsal surface with cyanoacrylic glue and a backpack harness style method (Van Winkel and Ji, 2014). As surgery often results in high mortality and direct attachments often become detached (Salmon, 2002), backpack harnesses are being adopted more often, as they appear to have no significant effect on behaviour, movement, or health of the individuals after a month of use (Van Winkel and Ji, 2014).

Mainland sites are more challenging for conservation management than offshore islands because they typically support species-rich predator assemblages that are impossible to eradicate and difficult to control. For example, in New Zealand, predator control on mainland sites have not been successful (Tocher, 2006; Hoare et al., 2007). Large-scale, full predator-proof enclosures and more effective predator control, although costly, may be the only remaining options for in situ management of threatened skinks at such sites. Both strategies have recently been successfully used to enhance survival of critically endangered Grand (*O. grande*) and Otago (*O. otagense*) skinks at Macraes Flat, New Zealand (Reardon et al., 2012), and could be developed on a smaller scale within Bermuda's nature reserves. Therefore, conservation effort should initially be directed to those islands where skinks have most recently been recorded within Castle Harbour (Inner Pear Rock, Nonsuch Island, Southampton Island, Castle Island, Horn Rock, Charles Island and Rushy Island) and would provide the best hope for continued survival of the skink population. Ensuring viable populations on numerous islands will help to provide a safety net in the event of a tropical storm or hurricane in the area.

A direct result of removing invasive predators is increased survival rates, population sizes and habitat usage in a variety of lizard species (Case, 1983; Case and Bolger, 1991; Newman, 1994; Towns, 1995; Castellano and Valone, 2006; Lettink et al., 2010; Reardon et al., 2012; McCreless et al., 2016). As invasive species pose a threat to Bermuda's biodiversity and are probably the main cause for skink declines, control measures should be a priority, especially on the smaller offshore islands. Due to an increase in cat feeding stations and abandonment of un-neutered pets, the only places the skinks remain are in cat-free protected areas or offshore islands. Indeed, there is a higher probability of being able to eradicate invasive species successfully on islands, compared to mainland sites (Clout and Veitch, 2002). A feral dog problem was resolved in Bermuda in the 1970s. This used an adoption or euthanasia programme, and a similar approach may reduce feral cats in areas in close proximity to protected areas. This is particularly the case at Spittal Pond and Coopers Island, where the illegal feeding of feral cats has been observed and also encourages the presence of rats and feral chickens (Bermuda Audubon Society, 2015). There are many other cost-effective measures to reduce predators. The use of rat bait should be re-evaluated due to evidence of secondary poisoning within the environment. Alternative methods could be used alongside bait to reduce its use, such as systematic trapping grids (Howald et al., 2009), or the management of natural predators of rats using barn owls (*Tyto alba)* of which a small naturally colonised population already exists in Bermuda (Maderios, 1991). Removing pesticides from the environment, providing nest boxes and supplementary feeding, has shown to increase barn owl breeding populations (Marti et al., 1979). As rats and mice make up 85% of Barn owl diet in Bermuda and Antiguan anoles are also a significant prey item during the summer months, the value of Barn owls as a pest control agent in Bermuda in considerable, especially as a breeding pair can consume more than 3,000 small mammals annually (Maderios, 1991). Deterrents could also be used to reduce invasive bird presence by removing suitable roosting and nesting sites by shortening and thinning dense forest habitat, coating eggs with liquid paraffin (Baker et al., 1993), and playing recorded distress calls of the birds (Gorenzel and Salmon, 1993).

As 96% of all skink captures during this study were found in the east of the Island within Castle Harbour, the remaining populations are vulnerable to natural disasters in the area. Translocations could alleviate the risk of extinction through the establishment of subpopulations in other less threatened locations. However, translocations are only feasible with sufficient detailed studies, background research and knowledge of the biology of wild populations to ensure sufficient resources are available year round and that long term effects on the species are considered (Conant, 1988; Dodd and Seigel, 1991; IUCN, 1996).

Additionally, future translocations may help to establish new sites outside the skink's natural range. Trunk Island (located within the Harrington Sound) has also been considered by the Department of Environment and Natural Resources as a release site due to being more sheltered from tropical storms within the Sound. Furthermore, the island has a dock that would allow easy access to the site and more frequent monitoring. However, continuous human presence on the island is equally likely to disturb the skinks and, unless bio secure procedures are investigated, invasive species have a high chance of being transferred from the mainland. There are other islands that could potentially support a viable skink population. Indeed, those lacking a dock or beach would deter landings, and reduce anthropogenic disturbance. Suitability of sites for reintroduction or translocation should there be determined by ensuring: (1) presence of breeding seabird populations; (2) absence of non-native terrestrial predators (i.e. cats, rats, toads, anoles); (3) absence of anthropogenic activities;, and (4) presence of low lying coastal vegetation, such as baygrape trees (*Coccoloba uvifera*), buttonwood (*Conocarpus erectus*), sea oxeye (*Borrichia frutescens*) and prickly pears (*Opuntia dillenii*). Inner Pear Rock and Palm Island are both small islands (< 0.4 hectares) already supporting skinks, and other small island sites that do not need significant habitat management and are unlikely to visited by tourists may also have potential (Pluess et al., 2012). Intensive skink surveys would not necessarily need to be undertaken regularly at every site. Some sub-populations may experience extinction and recolonisation while the metapopulation as a whole remains relatively stable. These dynamics need to be considered when understanding the findings from the skink surveys on Bermuda. Although anthropogenic structures and urban areas dominate the landscape – coastlines, dunes, and protected nature reserves provide the mix of suitable habitats within which the metapopulation can function, providing there is connectivity between them.

Although a monitoring programme may not need surveys every year, too long a gap between repeated surveys carries risks. Although natural populations fluctuate in size, they do not usually undergo severe reductions, followed immediately by restoration to the original population size. Rather, there is a gradual increase in population size (Maruyama and Fuerst, 1984). Therefore, if the population size becomes too low it restricts population growth potential, resulting in a population 'bottleneck' which reduces genetic variability and can be difficult to recover from, especially for small populations (Westemeier, 1998). For example, on Charles Island skinks were last studied there in 1998 and the island was found to have a healthy population (Raine, 1998). Sixteen years later multiple surveys did not detect skinks. Although hurricane Fabian hit the island in 2003 (a powerful category 4 storm), it is unclear if this was the cause of their apparent extirpation. There are many other examples of conservation initiatives around the world where studies show the population was observed to be declining (i.e. Monteverde harlequin frog *Atelopus* sp. and the golden toad *Bufo periglenes)* but extinction occurred as action was not taken soon enough (Crump et al., 1992; Pounds et al., 2006). Ideally, conservationists should allow populations to naturally recover following habitat and/or invasive species management. However, if stochastic events (e.g. hurricanes) alter the underlying dynamics of recovery, further interventions – such as translocations or reintroductions – may be desirable. Likewise, if population numbers remain are low, translocating individuals from larger, genetically diverse populations would help to increase fitness in smaller populations (Westemeier, 1998). Collecting remaining individuals for ex-situ conservation may also be beneficial by removing part of the population from a threatened habitat and placing it in a new location, which may be either in-situ or ex-situ. (IUCN, 2014).

### *6.1.5 Conclusion*

The Bermuda skink represents the country's only endemic terrestrial vertebrate, an extant species preserving millions of years of unique evolutionary history, and a high-profile flagship species for Bermuda and indicator of coastal environmental health. Within 370 years of human presence, skink populations have declined and have been extirpated from much of these formerly pristine islands and coastlines. This is due to the growing number of threats from habitat loss and destruction, increased predation, rapidly expanding coastal developments, lethal litter and anthropogenic activities (Davenport et al., 1997; Raine, 1998; Wingate, 1998; Glasspool and Outerbridge, 2004; Bacon et al., 2006; Edgar et al., 2010).

This study contributes to our understanding of skink population ecology and distribution and identifies critical drivers of declines and extinctions. Paradoxically, because of their isolation, the island populations are more viable than mainland populations that have unprecedented pressure from disturbance, development, and invasive species. Nevertheless, the island skink populations are still extremely susceptible to stochastic events such as hurricanes and extreme weather. This study provides recommendations for the development of effective long-term conservation management plans, with prioritised actions including:

(1) The continuation of island-wide population monitoring, including the use of PIT tagging individuals that will provide long-term data giving further insight into population trends, habitat use and movements between and within sites.

(2) The removal of non-native and forest habitat that is unsuitable for skinks, especially on small islands.

(3) Predator control, predator exclusion, and possibly some natural or artificial refuge supplementation (Lettink et al. 2010), which will allow skinks to escape predation.

Undertaking these actions is a crucial step towards protecting these elusive and endangered lizards from complete extinction as well as conserving their unique ecosystem.

## 6.1 LITERATURE CITED

Allen, C.R., Rice, K.G., Wojcik, D.P., and Percival, H.F. (1997). Effect of red imported fire ant envenomization on neonatal American alligators. *Journal of Herpetology*, 31:318–321.

ASIH. (2004). Guidelines for use of live amphibians and reptiles in field and laboratory research. Second Edition. Revised by the Herpetological Animal Care and Use Committee (HACC). American Society of Ichthyologists and Herpetologists: Lawrence, United States. Pp 43.

Bacon, J.P., Gray, J.A., and Kitson, L. (2006). Status and conservation of the reptiles and amphibians of the Bermuda Islands. *Applied Herpetology*, 3:323–344.

Baker, S.J., Feare, C. J., Wilson, C.J., Malam, D.S., and Sellars, G.R. (1993). Prevention of breeding of Canada geese by coating eggs with liquid paraffin. *International Journal of Pest Management* , 39:246–249.

Bermuda Audubon Society (2015). *Feral cats in Bermuda: a problem that can be solved*. [Online]. Available at: www.royalgazette.com/article/20150213/COMMENT/150219849 [Assessed 28 June 2019].

Blaustein, A.R., Walls, S.C., Bancroft, B.A., Lawler, J.J., Searle, C.L., and Gervasi, S.S. (2010). Direct and indirect effects of climate change on amphibian populations. *Diversity*, 2:281–313.

Bonney, R., Shirk, J.L., Phillips, T.B., Wiggins, A., Ballard, H.L., Miller-Rushing, A.J., and Parrish, J.K (2014). Next steps for citizen science. *Science*, 343.

Brandley, M.C., Wang, Y., Guo, X., Montes de Oca, A.N., Ortiz, M.F., Hikida, T., and Ota, H. (2010). Bermuda as an evolutionary life raft for an ancient lineage of endangered lizards. *PloS ONE*, 5: e11375.

Burton, A.C., Neilson, E., Moreira, D., Ladle, A., Steenweg, R., Fisher, J.T., Bayne, E., Boutin, S., and Stephens, P. (2015). Review: Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. *Journal of Applied Ecology*, 52:675–685.

Case, T.J. (1983). The reptiles: ecology. In: (Case TJ., and Cody ML. (Eds.) *Island Biogeography in the Sea of Cortez*. Pp 159–209. University of California Press: Berkeley, California, United States.

Case, T.J., and Bolger, D.T. (1991). The role of introduced species in shaping the distribution and abundance of island reptiles. *Evolutionary Ecology*, 5:272.

Castellano, M.J., and Valone, T.J. (2006). Effects of livestock removal and perennial grass recovery on the lizards of a desertified arid grassland. *Journal of Arid Environments*, 66:87–95.

Crump, M.L., Hensley, F.R., and Clark, K.L. (1992). Apparent decline of the golden toad: underground or extinct? *Copeia,* 2:413–420.

Conant, S. (1988). Saving endangered species by translocation: are we tinkering with evolution? *BioScience*, 38:254–257.

Davenport, J., Hills, J., Glasspool, A., and Ward, J. (1997). *A study of populations of the Bermudian rock lizard (skink), Eumeces longirostris, cope* (1861), *on the islands of Nonsuch and Southampton, Bermuda*. Department of Agriculture and Fisheries: Bermuda.

De Brooke, M.L, and Houston, D.C. (1983). The biology and biomass of the skinks *Mabuya sechellensis* and *Mabuya wrightii* on Cousin Island, Seychelles (Reptilia: Scincidae). *Journal of Zoology,* 200:179–195.

Dodd, K.C. (2016). *Reptile ecology and conservation: A handbook of techniques*. Oxford University Press: United Kingdom.

Doney, S.C., Fabry, V.J., Feely, R.A., and Kleypas, J.A. (2016). Ocean acidification: The other CO2 problem? *Washington Journal of Environmental Law & Policy*, 6:212 –251.

Edgar, P., Kitson, L., and Glasspool, A. (2010). *Recovery plan for the Bermuda skink, Eumeces longirostris*. Department of Environment and Natural Resources: Bermuda.

Fischlin, A., Midgley, G.F., Price, J.T., Leemans, R., Gopal, B., Turley, C., Rounsevell, M.D.A., Dube, O.P., Tarazona, J., and Velichko, A.A. (2007). "Ecosystems, their properties, goods, and services." In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ and Hanson CE (Eds.), *Climate change 2007: Impacts, adaptation and vulnerability*. Cambridge University Press, Cambridge, UK. Pp 211–272.

Frankham, R., Ballou, J.D., and Briscoe, D.A. (2002). *Introduction to conservation genetics*. Cambridge University Press: United Kingdom. Pp 617.

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

Goin, J.W. (1992). Requiem or recovery? *Texas Parks and Wildlife,* August: 28–35.

Gorenzel, W.P and Salmon, T.P. (1992). Urban crow roosts in California. *Proceedings of the Vertebrate Pest Conference,* 15:97–102.

Gorenzel, W.P., and Salmon, T.P. (1993). Tape-recorded calls disperse American crows from urban roosts. *Wildlife Society Bulletin*, 21:334-338.

Hoare, J.M., Adams, L.K., Bull, L.S., and Towns, D.R. (2007). Attempting to manage complex predator-prey interactions fails to avert imminent extinction of a threatened New Zealand skink population. *Journal of Wildlife Management*, 71:1576–1584.

Hoegh-Guldberg, O., Mumby, P.J., Hooten, A.J., Steneck, R.S., Greenfield, P., Gomez, E., Harvell, D.C., Sale, P.F., Edwards. A.J., Caldeira, K., Knowlton, N., Eakin, C.M., Iglesias-Prieto, R. (2007). Coral reefs under rapid climate change and ocean acidification. *Science*, 1737–1742.

Howald, G., Donlan, C.J., Faulkner, K.R., Ortega, S., Gellerman, H., Croll, DA., and Tershy, B.R. (2009). Eradication of black rats *Rattus rattus* from Anacapa island. *Oryx*, 44:30–40.

IUCN. (2014). *Guidelines on the use of ex situ management for species conservation*. version 2.0. Gland, Switzerland: IUCN Species Survival Commission. [Online]. Available at: www.iucn.org/about/work/programmes/species/publications/iucn_guidelines_and__policy__statements/Cove [Assessed 12 May 2019].

IUCN. (2017). *Guidelines for using the IUCN red list categories and criteria*, version.13. Prepared by the standards and petitions of subcommittee.

Jepsen, N., Schreck, C., Clements, S., and Thorstad, E. A brief discussion on the 2% tag/bodymass rule of thumb. In: Spedicato MT., Lembo G., and Marmulla G (Eds.) *Aquatic Telemetry: Advances and Applications – Proceedings of the Fifth Conference on Fish Telemetry: 9–13 June 2005; Ustica, Italy*, Rome: FAO; 2005:255–259.

Keller, L.F., and Waller, D.M. (2002). Inbreeding effects in wild populations. *Trends in Ecology and Evolution*, 17.

Koh, L.P., and Wich, S.A. (2012). Dawn of drone ecology: low-cost autonomous aerial vehicles for conservation. *Tropical Conservation Science*, 5:121–132.

Langkilde, T. (2009). Invasive fire ants alter behaviour and morphology of native lizards. *Ecology*, 90:208–217.

Lettink, M., Norbury, G., Cree, A., Seddon, P.J., Duncan, R.P., Schwarz, C.J. (2010). Removal of introduced predators, but not artificial refuge supplementation, increases skink survival in coastal duneland. *Biological Conservation,* 143:72–77.

Maderios, J.L. (1991). *Facts on the Barn Owl in Bermuda.* [Online] Available at: www.audubon.bm/birding/bermuda-birds/120-barn-owl [Assessed 28 June 2018].

Marti, C.D., Wagner, P.W., and Denne, K.W. (1979). Nest boxes for the management of Barn owls. *Wildlife Society Bulletin*, 7:145–148.

Maruyama, T., and Fuerst, P.A. (1984). Population bottlenecks and nonequilibrium models in population genetics. I. Allele numbers when populations evolve from zero variability. *Genetics*, 108:745-763.

McCreless, E.E., Huff, D.D., Croll, D.A., Tershy, B.R., Spatz, D.R., Holmes, N.D., Butchart, S.H.M., and Wilcox, C. (2016) . Past and estimated future impact of invasive alien mammals on insular threatened vertebrate populations. *Nature Communications*, 7:12488.

Miller, K.A., Nelson, N.J., Smith, H.G., and Moore J.A. (2009). How do reproductive skew and founder group size affect genetic diversity in reintroduced populations? *Molecular Ecology*, 18:3792–3802.

Mooney, H.A., and Cleland, E.E. (2001). The evolutionary impact of invasive species. *Proceedings of the National Academy of Sciences*, 98: 5446–5451.

Moulis, R.A. (1997). Predation by the imported fire ant (*Solenopsis invicta*) on loggerhead sea turtle (*Caretta caretta*) nests on Wassaw national wildlife refuge, Georgia. *Chelonian Conservation Biology*, 2:433–436.

Newman, D.G. (1994). Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, *Cyclodina macgregori*. *New Zealand Journal of Zoology*, 21:443-456.

Olsen, S.L., Hearty, P.J., and Pregill, G.K. (2006). Geological constraints on evolution and survival in endemic reptiles on Bermuda. *Journal of Herpetology,* 40:394–398.

Pike, D.A., Pizzatto, L., Pike, B.A., and Shine, R. (2008). Estimating survival rates of uncatchable animals: the myth of high juvenile mortality in reptiles. *Ecology*, 89:607–611.

Pounds, J.A., Bustamante, M.R., Coloma, L.A., Consuegra, J.A., Fogden, M.P.L *et al.* (2006). Widespread amphibian extinctions from epidemic disease driven by global warming. *Nature,* 439:161–167.

Pluess, T., Cannon, R., Jarosık, V., Pergl, J., Pysek, P., and Bacher, S. (2012). When are eradication campaigns successful? A test of common assumptions. *Biological Invasions*, 14:1365–1378.

Raine, A. (1998). *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated populations of the critically endangered Bermuda Rock Lizard (Eumeces longirostris).* MSc Thesis. University College London: London, United Kingdom.

Reardon, J.T., Whitmore, N., Holmes, K.M., Judd, L.M., Hutcheon, A.D., Norbury, G., and Mackenzie, D.I. (2012). Predator control allows critically endangered lizards to recover on mainland New Zealand. *New Zealand Journal of Ecology*, 36:141–150.

Rowcliffe, J.M., Field, J., Turvey, S.T., and Carbone, C. (2008). Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology*, 45:1228–1236.

Saccheri, I., Kuussaari, M., Kankare, M., Vikman, P., Fortelius, W., and Hanski, I. (1998). Inbreeding and extinction in a butterfly metapopulation. *Nature*, 392:491–494.

Salmon, N.M. (2002). Telemetric studies in geckos *Hoplodactylus maculatus* and *Naultinus gemmeus*. MSc Thesis. University of Otago, Dunedin, New Zealand.

Sayles, R.W. (1931). Bermuda during the Ice Age. *Proceedings of the American academy of arts and sciences*, 66:381–468.

Smircich, M.G., and Kelly, J.T. (2014). Extending the 2% rule: the effects of heavy internal tags on stress physiology, swimming performance, and growth in brook trout. *Animal Biotelemetry*, 2:16.

Templeton, A.R., Robertson, R.J., Brisson, J., and Strasburg, J. (2001). Disrupting evolutionary processes: The effect of habitat fragmentation on collared lizards in the Missouri Ozarks. *PNAS*, 98:5426–5432.

Tocher, M.D. (2006). Survival of Grand and Otago skinks following predator control. *The Journal of Wildlife Management,* 70:31–42.

Towns, D.R. (1995). Changes in habitat use by lizards on a New Zealand island following removal of the introduced Pacific Rat *Rattus exulans*. *Pacific Conservation Biology*, 2:286–292.

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

Van Winkel, D and Ji, W. (2014). Attaching radio-transmitters to geckos: Trials and tribulations. *Herpetological Review*, 45:13 –17.

Verrill, A.E. (1902). *The Bermuda islands.* Transactions of the Connecticut Academy of Arts and Sciences*l:* New Haven: Connecticut, United States, 11:38.

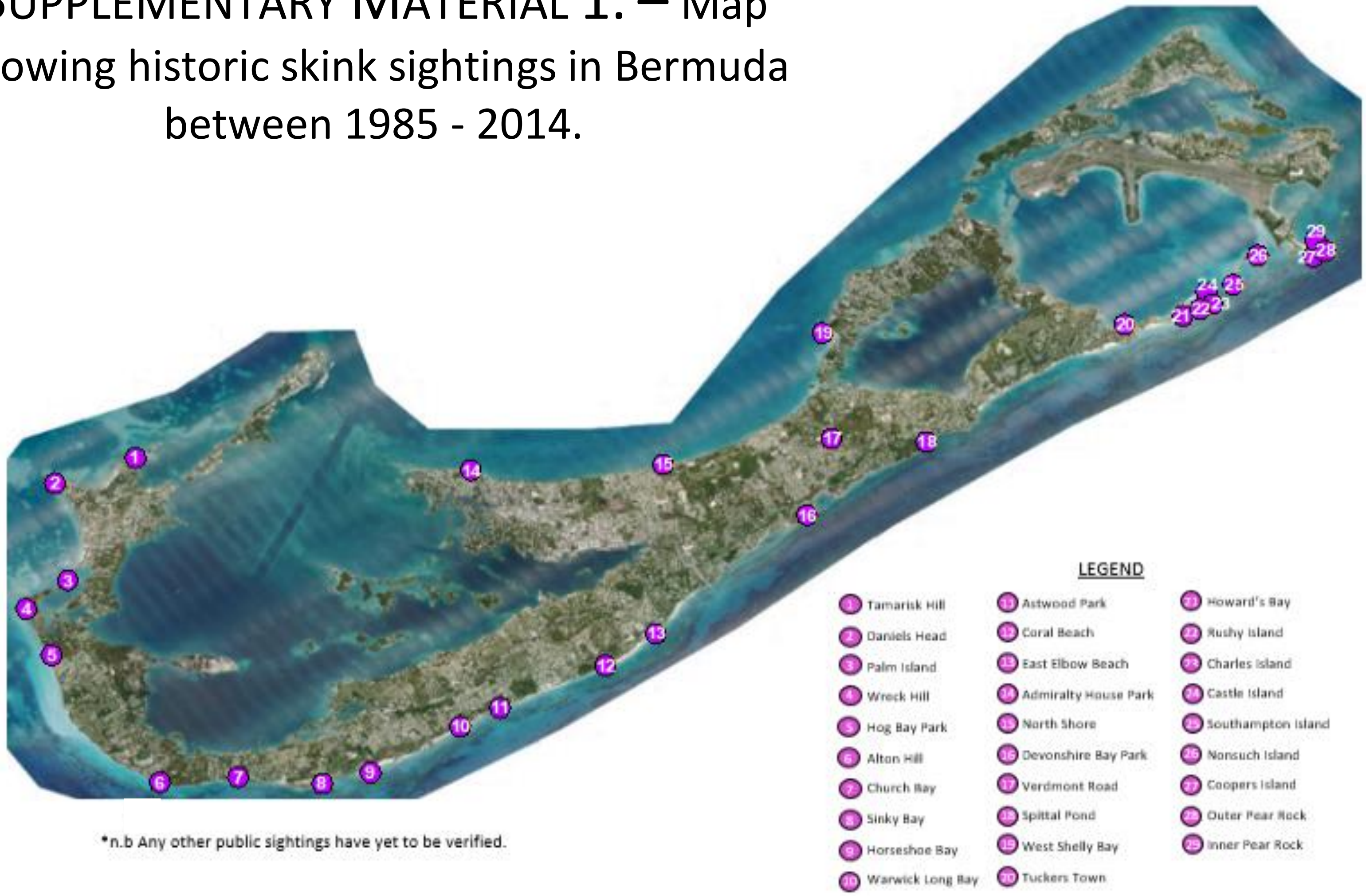
Wasser, S.K., Hayward, L.S., Hartman, J., Booth, R.K., Broms, K., Berg, J., Seely, E., Lewis, L., and Smith, H. (2012). Using detection dogs to conduct simultaneous surveys of Northern spotted (*Strix occidentalis caurina*) and Barred owls (*Strix varia*). *PLoS ONE,* 7: e42892.

Williams, R.L., Singleton, G.R., and Dickman, C.R. (2003). “The role of interspecific competition in determining macrohabitat use by the black rat and brown rat at Bradley’s Head, NSW.” In: Singleton GR, Hinds LA, Krebs CA and Spratt DM (Eds.), *Rats, mice and people: rodent biology and management*. ACIAR Monograph: Canberra. Pp 564.

Welch, D.W., Batten, S.D., and Ward, B.R. (2007). Growth, survival, and tag retention of steelhead trout (*O. mykiss*) surgically implanted with dummy acoustic tags. *Hydrobiologia*, 582:289.

Westemeier, R.L., Brawn, J.D., Simpson, S.A., Esker, T.L., Jansen, R.W., Walk, J.W., Kershner, E.L., Bouzat, J.L., Paige, K.N. (1998). Tracking the long-term decline and recovery of an isolated population. *Science*, 282:1695–1697.

# SUPPLEMENTARY MATERIAL 1. – Map showing historic skink sightings in Bermuda between 1985 - 2014.



*n.b Any other public sightings have yet to be verified.

# SUPPLEMENTARY MATERIAL 2. – Summarised data from previously published and unpublished *Plestiodon longirostris* surveys.

| Location | No. of Skinks Captured | Survey year | Source |
|---|---|---|---|
| Abbots Cliff Park | 0 | 2001 | Kitson, L |
| | 0 | 2003 | Kitson, L |
| Admiralty Park | 0 | 2002 | Kitson, L |
| | 0 | 2003 | Kitson, L |
| Alton Hill | 1 | 2003 | Kitson, L |
| Astwood Park | 1 | 2001 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Baileys Bay | 0 | 2003 | Kitson, L |
| Castle Island | 22 | 1993 | Conyers, J |
| | 39 | 1998 | Wingate, R |
| | 21 | 2000 | Hammond, M |
| | - | 2002 | Kitson, L |
| | 150 | 2015 | Turner, H |
| | 121 | 2016 | Turner, H |
| | 56 | 2017 | Turner, H |
| Charles Island | 62 | 1998 | Raine, A |
| | 0 | 2016 | Turner, H |
| | 0 | 2017 | Turner, H |
| Church Bay | 1 | 2002 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Coney Island | 0 | 2001 | Kitson, L |
| | 0 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Coopers Island | 2 | 2013 | Outerbridge, M & Garcia, G |
| | 2 | 2016 | Turner, H |
| | 0 | 2017 | Turner, H |
| Coral Beach Reserve | 0 | 2002 | Kitson, L |
| | 1 | 2003 | Kitson, L |
| Daniels Head Point | 1 | 2002 | Kitson, L |
| | 2* | 2015 | Turner, H & Outerbridge, O |
| Devonshire Bay | 0 | 2002 | Kitson, L |
| | 1 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Duckling Stool Park | 0 | 2002 | Kitson, L |
| Ferry Point Park | 0 | 2001 | Kitson, L |
| | 0 | 2003 | Turner, H |
| | 0 | 2016 | Turner, H |
| Fort St Catherine | 0 | 2002 | Kitson, L |
| Gilbert Hill | 1 | 2002 | Kitson, L |
| Great Head Park | 0 | 2002 | Kitson, L |
| | 0 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |

| | | | |
|---|---|---|---|
| Green Rock | 0 | 1998 | Raine, A |
| | 0 | 2017 | Turner, H |
| Hog Bay Park | 2 | 2001 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Horn Rock | - | 1991 | Griffith, H et al., |
| | 33 | 2011 | Glasspool, A & Ward, J |
| | 0 | 2016 | Turner, H |
| Horseshoe Bay | 1 | 2001 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Howard Bay | 1 | 2002 | Kitson, L |
| Hungry Bay | 0 | 2000 | Kitson, L |
| | 0 | 2003 | Kitson, L |
| Inner Pear Rock | 37 | 1998 | Raine, A |
| | 2 | 2016 | Turner, H |
| | 0 | 2017 | Turner, H |
| Ireland Island South | 0 | 2003 | Kitson, L |
| Little Head Park | 0 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Long Rock | 0 | 1998 | Raine, A |
| | 0 | 2016 | Turner, H |
| Nonsuch Island | - | 1991 | Griffith, H et al., |
| | 23 | 1997 | Davenport, J |
| | - | 2002 | Kitson, L |
| | 41 | 2015 | Turner, H |
| | 17 | 2016 | Turner, H |
| | 39 | 2017 | Turner, H |
| North Cock Rock | 2 | 2016 | Turner, H |
| North Shore | 1 | 2002 | Kitson, L |
| | 0 | 2015 | Turner, H & Outerbridge, O |
| Outer Pear Island | 12 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Palm Island | 44 | 1998 | Wingate, R |
| | 2 | 2013 | Outerbridge, M & Garcia, G |
| | 1 | 2015 | Turner, H |
| Rushy Island | 14 | 2002 | Kitson, L |
| | 1 | 2016 | Turner, H |
| Scaur Hill Fort | 0 | 2001 | Kitson, L |
| | 0 | 2002 | Kitson, L |
| | 0 | 2016 | Turner, H |
| Sinky Bay | 0* | 2015 | Turner, H |
| Southampton Island | 115 | 1997 | Davenport, J |
| | - | 2002 | Kitson, L |
| | 277 | 2004 | Glasspool, A & Outerbridge, M |
| | 36 | 2013 | Outerbridge, M & Garcia, G |
| | 125 | 2015 | Turner, H |
| | 370 | 2016 | Turner, H |
| | 197 | 2017 | Turner, H |
| South Cock Rock | 1 | 2016 | Turner, H |
| Spittal Pond | 54 | 1998 | Raine, A |
| | 15 | 2013 | Outerbridge, M & Garcia, G |
| | 21 | 2015 | Turner, H |

| | | | |
|---|---|---|---|
| | 13 | 2016 | Turner, H |
| | 11 | 2017 | Turner, H |
| Tamarisk Hill | 1 | 2002 | Kitson, L |
| Tuckers Town | 1 | 2001 | Kitson, L |
| Warwick Long Bay | 1 | 2003 | Kitson, L |
| | 0 | 2016 | Turner, H |
| West Shelley Bay | 1 | 2003 | Kitson, L |
| | 0 | 2015 | Turner, H |
| Wreck Hill Estate | 1 | 2001 | Kitson, L |

* Indicates skink remains were also found at the site.

*n.b A* hyphen(-) represents an unknown number of captures.

# SUPPLEMENTARY MATERIAL 3. – Marking Method.

For population monitoring studies, lizards are almost always assigned unique identifying marks. In short term monitoring studies, quick drying enamel paint or coloured tape are frequently applied in order to mark individuals (Blair, 1960; Dunham, 1981; Glasspool and Outerbridge, 2004; Shine and Olsson, 1998). However, these methods often wear off within a few days, disappear when the skink sloughs it skin and any external conspicuous markers may increase predation risk. Therefore, for monitoring studies over longer periods a variety of techniques have been used in order to mark lizards more permanently. Heat-branding has occasionally been used to mark lizards (Ekner et al. 2011). Tattooing has been used to mark snakes (Woodbury, 1948) and could be applied to lizards. However, one of the most commonly used methods includes toe-clipping (Bellairs and Bryant, 1968) where the distal phalanx of one or more digits are amputated to a coded pattern (Barwick, 1959; Blair, 1960; Brooks, 1967; Ballinger, 1973; Ruibal and Philibosian, 1974; Schoener and Schoener, 1982; Ruby and Dunham, 1984; Tinkle and Dunham, 1986). However, it has been termed an invasive identification method in some studies that have shown to negatively affect survival or behaviour in lizards (Schmidt and Schwarzkopf, 2010; Parris and McCarthy, 2001; McCarthy and Parris, 2004). Nevertheless, some studies show the procedure causes little trauma and there are no negative effects (Huey et al. 1990; Ott and Scott, 1999; Paulissen and Meyer, 2000; Borges-Landaez and Shine, 2003). Nevertheless, it is not necessarily an accurate technique as toe loss naturally occurs quite frequently, particularly in skinks (Hudson, 1996). As a result, there would be a high chance of misidentification of individuals marked by toe-clipping (Rand, 1965; Schoener and Schoener, 1980; Hudson, 1996) especially during long term monitoring.

Passive integrated transponders (PIT tags) are a novel tagging method. Individuals can be tagged quickly and easily in the field, it prevents misidentification and will be permanent over their lifetime as the tags can be accurately read for as long as seventy-five years (Biomark, 2015; Ott and Scott, 1999). The PIT tags are glass encased electromagnetic coils with a microchip that is implanted under the skin or into the body cavity. If a PIT tag is present, the reader generates a close-range, low electromagnetic field that immediately activates the tag, which transmits its unique number back to the reader (Camper and Dixon, 1988) therefore it is not battery operated and can always be read by the PIT tag reader therefore handling the lizards in the future will be minimal and could reduce the chances of caudal autonomy.

Even though the application of the internal tags may induce temporary stress (Langkilde and Shine, 2006), this is of short duration and highly variable between individuals and if inserted properly the tags are not known to cause any significant negative impacts (Ferner, 1979) or affect growth rates or locomotor performance of lizards (Keck, 1994; Jemison et al. 1995). As a result of using PIT tags, individuals can be monitored more closely in the long term by looking at the species growth, population density, dispersal or immigration over time that all require a permanent marking method (Jemison et al. 1995; Ott and Scott, 1999) as well as reducing handling time and causing potential injury from other marking methods.

Previous studies involving Bermuda skink marking have used temporary marking techniques such as enamel paints (Glasspool and Outerbridge, 2004) and coloured tape (Shine and Olsson, 1998) that would not be suitable for this study as they would wear off within a few days, disappear when the skink sloughs it skin and any external conspicuous markers may increase predation risk. Therefore, skinks collected from the pit fall traps were implanted with a passive integrated transponder also known as a PIT tag (8 mm x 1.4 mm 25 mg ID162B FDX-B Trovan® www.trovan.com) operating at a frequency of 134.2 kHz. All tags were inserted with a syringe type implanter (IM-200) and 1.25" 14-gauge sterile disposable hypodermic needle subcutaneously in either the left or right lateral side of the body, as the correct placement can reduce tag damage and loss (Germano and Williams, 1993). Prior to the injection, the sterile needle was loaded to the syringe and the injection site was wiped clean with diluted F10® antiseptic solution and the animal was positioned on its left lateral side. The needle was injected below a ventral scale (in the lower abdomen, to avoid all major organs) where the tag was pushed just below the skin (Fig. 2.3). Once implanted the tags were then read by passing a PIT tag reader (Universal (LID-560) scanner-Trovan®) within 34 cm of the animal which identified the individual’s unique identification code. Immediately after a drop of fast drying cyanoacrylate glue was applied over the injection site to prevent tag loss and speed wound healing (Gibbons and Andrews, 2004). Once individuals were marked, they were immediately released back into the population and resampled to see what fraction of the individuals carry marks. As a result of using PIT tags, individuals can be monitored more closely in the long term by looking at the species growth, population density, dispersal or immigration over time that all require a permanent marking method (Ott and Scott, 1999; Jemison et al. 1995) as well as reducing handling time and causing potential injury from other marking methods.

**FIGURE 2.3**. Using passive integrated transponder (PIT) tags to mark skinks for long-term monitoring (*Photo Credit: Turner, H. 2015*).

*Ethics statement*

This research was conducted under a ‘Level 1’ permit with consent from the Government of Bermuda Department of Conservation Services (DCS). As the Bermuda skink is categorised as a critically endangered species according to the IUCN (Conyers and Wingate, 1996) and is protected under the Protected Species Amendment Act (2011) all research was carried out in strict compliance with this legislation. An ethical review was also approved by the University of Kent, Chester Zoo and the Bermuda Aquarium, Museum and Zoo.

*Consideration to ethics*

The standard traps used are made from glass and covered in oil to attract skinks, this can cause them to reach high temperatures if left in direct sunlight (highest temperature record during this study was 70.84°C inside a glass jar). Trap mortality is therefore a serious consideration and every effort was put into place to prevent animals getting caught in traps and consequently becoming exhausted attempting to escape, becoming heat stressed due to the environmental conditions within the trap or being vulnerable to predation. Therefore, traps were checked at least once an hour and all traps closed off when not in use or become unmanageable, ensuring there are enough field assistants to do so. Traps were placed in shaded areas and where possible palm leaves were used to cover the top of the traps, to prevent overheating. All skinks caught were released as close to their point of capture, to give them the best chance of survival.

General disturbances in the survey area were minimised as much as possible by checking the traps at scheduled times, taking care not to trample vegetation and by taking rubbish with us, which could otherwise prevent the skinks performing normal behaviours such as basking and foraging. Rarely, non-focal species were captured such as cockroach nymphs, Jamaican anoles, brown rats, a hermit crab and a toadlet were recorded in the traps but were not found to cause any injury to the skinks and were all released immediately away from the study area.

## LITERATURE CITED

Ballinger, R.E. (1973). Comparative demography of two viviparous iguanid lizards (*Sceloporus jarrovi* and *Sceloporus poinsetti*). *Ecology*, 54:269–283.

Barwick, R.E. (1959). The life history of the common New Zealand skink *Leiolopisma zelandica* (Gray, 1843). *Transactions of the Royal Society of New Zealand*, 86:331–380.

Brooks, G.R., Jr. (1967). Population ecology of the ground skink, *Lygosoma laterale* (Say). *Ecological Monographs*, 37:72–87.

Bellairs, A.D., and Bryant, S.V. (1968). Effects of amputation of limbs and digits of lacertid lizards. *The Anatomical Record*, 161:489–495.

Biomark. (2015). *RFID and PIT technology explained*. [Online]. Available at: http://www.biomark.com/contact_us/faqs/ [Accessed 5 July 2015].

Blair, W.F. (1960). The rusty lizard: A population study. University of Texas Press: Austin, United States.

Borges-Landaez, P.A., and Shine, R. (2003). Influence of toe-clipping on running speed in *Eulamprus quoyii*, an Australian scincid lizard. *Journal of Herpetology*, 37:592–595.

Camper, J.D., and Dixon, J.R. (1988). Evaluation of a microchip marking system for amphibians and reptiles. *Texas Parks and Wildlife Department, Austin, Texas, Research Publication, 7100:*1–22.

Conyers, J., and Wingate, D. (1996). *Plestiodon longirostris. The IUCN red list of threatened species 1996.* [Online] Available at: http://dx.doi.org/10.2305/IUCN.UK.1996.RLTS.T8218A12900393.en [Assessed 23 August 2017].

Dunham, A.E. (1981). Populations in a fluctuating environment: The comparative population ecology of *Sceloporus merriami* and *Urosaurus ornatus*. *Miscellaneous Publications of the University of Michigan Museum of Zoology,* 158:1–62.

Ekner, A., Sajkowska, Z., Dudek, K., and Tryjanowski, P. (2011). Medical cautery units as a permanent and non-invasive method of marking lizards. *Acta Herpetologica*, 6:229–236.

Ferner, J.W. (1979). A review of marking techniques for amphibians and reptiles. *SSAR Herpetological,* 9:41.

Germano, D., and Williams, D. (1993). Field evaluation of using passive integrated transponder (PIT) tags to permanently mark lizards. *Herpetological Review,* 24:54–56.

Gibbons, J.W., and Andrews, K.M. (2004). PIT tagging: simple technology at its best. *Bioscience*, 54:447–454.

Glasspool, A., and Outerbridge, M. (2004). *A population re-survey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island, Castle Harbour.* Bermuda Zoological Society: Bermuda.

Hudson, S. (1996). Natural toe loss in southeastern Australia skinks: implications for marking lizards by toe-clipping. *Journal of Herpetology,* 30:106–110.

Huey, R.B., Dunham, A.E., Overall, K.L., and Newman, R.A. (1990). Variation in locomotor performance in demographically known populations of the lizard *Sceloporus merriami*. *Physiological and Biochemical Zoology,* 63.

Jemison, S.C., Bishop, L.A., May, P.G., and Farrell, T.M. (1995). The impact of PIT tags on growth and movement of the rattlesnake *Sistrurus miliarius*. *Journal of Herpetology,* 29:129–132.

Keck, M.B. (1994). Test for detrimental effects of PIT tags in neonatal snakes. 1994:226–228.

Langkilde, T., and Shine, R. (2006). How much stress do researchers inflict on their study animals? A case study using a scincid lizard, *Eulamprus heatwolei. The Journal of Experimental Biology*, 209:1035–1043.

McCarthy, M.A., and Parris, K.M. (2004). Clarifying the effect of toe clipping on frogs with bayesian statistics. *Journal of Applied Ecology*, 41:780–786.

Ott, J., and Scott, D. (1999). Effects of toe clipping and PIT tagging on growth and survival in metamorphic *Ambystoma opacum*. *Journal of Herpetology,* 33:348–351.

Paulissen, M.A., and Meyer, H.A. (2000). The effect of toe clipping on the gecko *Hemidactylus turcicus. Journal of Herpetology*, 34:282–285.

Parris, K.M., and McCarthy, M.A. (2001). Identifying effects of toe clipping on anuran return rates: the importance of statistical power. *Amphibia-Reptilia*, 22:275–289.

Rand, A.S. (1965). On the frequency and extent of naturally occurring foot injuries in *Tropidurus torquatus* (Sauria, Iguanidae). *Papeis Avulsos de Zoologia,* 17:225–228.

Ruby, D.E., and Dunham, A.E. (1984). A population analysis of the ovoviviparous lizard *Sceloporus jarrovi* in the Pinaleño mountains of Southeastern Arizona. *Herpetologica*, 40:425–436.

Ruibal, R., and Philibosian, R. (1974) The population ecology of the lizard *Anolis acutus*. *Ecology*, 55:525–537.

Schmidt, K., and Schwarzkopf, L. (2010). Visible implant elastomer tagging and toe clipping: effects of marking on locomotor performance of frogs and skinks. *Journal of Herpetology,* 20:99–105.

Shine, R., and Olsson, M. (1998). Chemosensory mate recognition may facilitate prolonged mate guarding by male snow skinks, *Niveoscincus microlepidotus*. *Behavioural Ecology and Sociobiology,* 43:359–363.

Schoener, T.W., and Schoener, A. (1980). Densities, sex ratios, and population structure in four species of Bahamanian *Anolis* lizards. *Journal of Animal Ecology,* 49:19–53.

Schoener, T.W., and Schoener, A. (1982). The ecological correlates of survival in some Bahamanian *Anolis* lizards. *Oikos*, 39:1–16.

Tinkle, D.W., and Dunham, A.E. (1986). Comparative life histories of two syntopic *sceloporine* lizards. *Copeia*, 1986:1–18.

Woodbury, A.M. (1948). Marking reptiles with an electric tattooing outfit. *Copeia*, 1948:127–128.

# SUPPLEMENTARY MATERIAL 4. – Vegetation classification by habitat type.

| Habitat Type | Vegetation |
|---|---|
| Beach / Dune / Rocky Outcrops  | Coastal spurge (*Euphorbia mesembrianthemifolia*), Sea purslane (*Sesuvium portulacastrum*), Sea oxeye (*Borrichia arborescens*), Bay bean (*Canavalia rosea*), Seacoast bulrush (*Bolboschoenus robustus*), Sea purslane (*Sesuvium portulacastrum*), Seaside morning glory (*Ipomoea pes-caprae*), Beach-tea croton (*Croton punctatus*), Seaside golden rod (*Solidago sempervirens*), Seaside evening-primrose (*Oenothera humifusa*), Beach Lobelia/Inkberry (*Scaevola plumieri*), Beach Naupaka (*Scaevola sericea*). |
| Lowland Scrub  | Capeweed (*Lippia nodiflora*), Sea oxeye (*Borrichia arborescens*), Bermudiana (*Juniperus bermudiana*), Josephs coat (*Euphorbia (Poinsettia) heterophylla*), Erect prickly pear cactus (*Opuntia dillenii*), Plantain (*Plantago lanceolata*), West Indian grass (*Eustachys petraea*), Darrell's fleabane (*Erigeron darrellianus*), Black medick (*Medicago lupulina*), Seaside creeping daisy (*Wedelia trilobata*), Seaside golden rod (*Solidago sempervirens*), Iodine bush (*Mallatonia gnaphalodes*), Tassel plant (*Suriana maritima*), Sage bush/Lantana (*Lantana camara*), Turnera (*Tutnera ulmifolia*), St. Andrew's cross (*Hypericum hypericoides*), Bermuda Snowberry (*Chiococca alba*), White beggar's tick (*Bidens pilosa*), Common dandelion (*Taraxacum officinale*), St Augustine grass (*Stenotaphrum secundatum*), Crabgrass (*Digitaria* sanguinalis), Bermuda grass (*Cynodon dactylon*), Stinging nettles (*Urtica dioica*), Spiny sow thistle (*Sonchus asper*), Floppers (*Kalanchoe pinnata*). |
| Dense scrub / Coastal Forest  | Seaside golden rod (*Solidago sempervirens*), Prickly pear cactus (*Opuntia dillenii*), Scurvy grass (*Cakile lanceolata*) , **Asparagus fern (*Asparagus densiflorus*),** Fennel (*Foeniclum vulgare*), Coast sophora (*Sophora tomentosa*), Box briar (*Randia aculeata*), Poison ivy (*Toxicodendron radicans*), **Mexican petunia (*Ruellia brittoniana*)**, Golden mimosa (*Acacia dealbata*), Bay grape (*Coccoloba uvifera*),<br>**Tamarisk (*Tamarix gallica*),** Casuarina **(*Casuarina equisetifolia*)**, Bermuda palmetto (*Sabal bermudana*), Buttonwood (*Conocarpus erectus*), Bermuda cedar (*Juniperus bermudiana*), **Brazil pepper (*Schinus terebinthifolius*),** Spanish bayonet (*Yucca aloifolia*), Jamaica Dogwood (*Dodonaea vicosa*), Forestiera (*Forestiera segregata*). |

*Those highlighted in **bold** are considered an invasive, non-native species.

# SUPPLEMENTARY MATERIAL 5. – Publications

- Turner, H., Griffiths, R.A., Garcia, G., and Outerbridge, M. (2017). Natural History Notes: *Plestiodon longirostris* (Bermuda Skink), Tail Bifurcation. Herpetological Review, 48(1): 199.

- Turner, H. (2017). Geographic Distribution: *Plestiodon longirostris* (Bermuda Skink). Herpetological Review, 48(4): 812.

- Turner, H. (2018). Bermuda skink project. *The Natterjack*, 216: 9-11.

- Turner, H., Griffiths, R.A., Garcia, G., and Outerbridge, M. (2019). Estimating population parameters for the critically endangered Bermuda skink using robust design capture-mark-recapture modelling. *Oryx*, 18 doi:10.1017/S0030605318001485

# Estimating population parameters for the Critically Endangered Bermuda skink using robust design capture–mark–recapture modelling

Helena Turner, Richard A. Griffiths
Mark E. Outerbridge and Gerardo Garcia

**Abstract** Reliably estimating population parameters for highly secretive or rare animals is challenging. We report on the status of the two largest remaining populations of the Critically Endangered Bermuda skink *Plestiodon longirostris*, using a robust design capture–mark–recapture analysis. Skinks were tagged with passive integrated transponders on two islands and captured on 15 sampling occasions per year over 3 years. The models provided precise estimates of abundance, capture and survival probabilities and temporary emigration. We estimated skink abundance to be 547 ± SE 63.5 on Southampton Island and 277 ± SE 28.4 on Castle Island. The populations do not appear to be stable and fluctuated at both sites over the 3-year period. Although the populations on these two islands appear viable, the Bermuda skink faces population fluctuations and remains threatened by increasing anthropogenic activities, invasive species and habitat loss. We recommend these two populations for continued monitoring and conservation efforts.

**Keywords** Bermuda skink, capture–mark–recapture analysis, conservation, *Plestiodon longirostris*, population monitoring, reptile, robust design, survival

Supplementary material for this article is available at https://doi.org/10.1017/S0030605318001485

---

## Introduction

Many threatened species are cryptic, elusive and challenging to survey. The development of efficient survey methods to detect them is vitally important for conservation management. A wide variety of methods have been used in monitoring studies, including capture–mark–recapture techniques to estimate abundance and survival (Krebs, 1999; Besbeas et al., 2002), which are critical determinants of population viability (White & Burnham, 1999). Conventionally, marking is used to uniquely identify individuals in successive samples. For lizards, photographic identification using natural markings (Sacchi et al., 2010), tagging with passive integrated transponders (Germano & Williams, 1993) and individual recognition from DNA sampling (Moore et al., 2009) have increased the utility and application of the capture–mark–recapture approach.

The Bermuda skink *Plestiodon longirostris* is the country's only extant endemic reptile (Bacon et al., 2006). The species primarily feeds on terrestrial and leaf litter arthropods such as ants and woodlice (Wingate, 1965) but also on a variety of other food sources such as prickly pear cactus *Opuntia* sp. and fruits (Wingate, 1965; Davenport et al., 1997, 2001; Edgar et al., 2010). Unlike many other reptiles, the skinks are scavengers with a keen sense of smell that attracts them to carrion (Garber, 1988; Davenport et al., 2001), and will make use of seasonally abundant food sources such as broken eggs, dead chicks and dropped fish from nesting colonies of the native white-tailed tropicbird *Phaethon lepturus catesbyi* and the endemic Bermuda petrel *Pterodroma cahow* (Garber, 1988; Edgar et al., 2010).

The skinks were once common throughout the islands of Bermuda but populations have undergone significant declines since the 1990s when they were reportedly rarely seen on the mainland because of increased anthropogenic disturbance, habitat loss and degradation, and the introduction of invasive flora and fauna (Davenport et al., 1997; Glasspool & Outerbridge, 2004). The species is now restricted to the edges of rocky coastal habitat and is categorized as Critically Endangered on the IUCN Red List (Conyers & Wingate, 1996), with < 2,500 individuals thought to be left on Bermuda (Edgar et al., 2010; Turner, 2019). Recent surveys have confirmed skinks at only 14 sites within nature reserves and offshore islands (Turner, 2019), and the continued threats are likely to have a major impact on the remaining fragmented and isolated populations.

We used a robust design model to monitor trends in abundance, survival, capture probabilities and temporary emigration of the two largest skink populations on Bermuda. The robust design model of Pollock (1982) is an extension of the Cormack–Jolly–Seber model and has become increasingly popular as it combines the advantages

Helena Turner (Corresponding author, orcid.org/0000-0002-6067-888X) and Richard A. Griffiths (orcid.org/0000-0002-5533-1013) Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Canterbury, UK
E-mail hturnerjsy@hotmail.com

Mark E. Outerbridge Government of Bermuda, Department of Environment and Natural Resources, Flatt's Village, Bermuda

Gerardo Garcia (orcid.org/0000-0002-3007-8998) Chester Zoo, Cedar House, Chester, UK

*Received 28 July 2018. Revision requested 14 September 2018.*
*Accepted 6 December 2018.*

of the live-recapture model and the closed-capture models. The model and its assumptions are described in detail by Cormack (1968), Otis et al. (1978), Seber (1982, 1986) and Kendall et al. (1995). The current management plan for the Bermuda skink calls for monitoring every 3–5 years (Edgar et al., 2010) but the last capture–mark–recapture surveys were undertaken in 2004, on Southampton Island, and the current status of the population was unknown. More frequent estimates of population parameters will be necessary to inform conservation management and future monitoring programmes.

## Study area

This study was undertaken at two sites in Castle Harbour, Bermuda: the 0.8 ha Southampton Island and 1.4 ha Castle Island (Fig. 1). We chose these two sites because they are thought to harbour the largest known Bermuda skink populations, are both within protected nature reserves and are considered the most suitable areas for targeted conservation efforts. The islands are separated by a 330 m wide water channel and their abiotic factors are similar. Castle Island is closer to the main island (164 m) and faces more threats from invasive species and anthropogenic activities, which have been the main reasons for the skink's decline at this location. We surveyed additional sites across Bermuda but sample sizes were too small (< 50 individuals) to provide reliable estimates of abundance.

## Methods

### Skink capture and marking

We adapted methods from Davenport et al. (1997) and used a consistent survey protocol across the two locations. We used 65 glass jars (c. 240 × 160 mm, volume 4.5 l) as traps, set up in 10 × 10 m grid networks (skink home ranges are c. 10 $m^2$; Davenport et al., 1997). Traps were mounted at a 45–90° angle, with rocks or vegetation placed at the opening so skinks could gain access. We used flagging tape to label each trap, recorded locations with a handheld GPS and used dried palmetto *Sabal bermudana* fronds and small towels to shade the traps. The pitfall traps were baited daily with 10 ml of canned sardines, which were placed in a small sealed tea strainer to prevent consumption, which could affect recapture rates. We smeared 5 ml of cod liver oil around the rim of each trap to prevent skinks escaping and to deter ants (Davenport et al., 1997). Traps were checked hourly during 11.00–16.00 and were then closed by removing the bait and turning the jar upside down to prevent accidental captures. We did not carry out surveys at temperatures < 21 °C or when heavy rainfall or winds (> 25 km/h) were forecast because the lizards are less active under such conditions (M. Outerbridge, 2015, pers. comm.) and because reaching the islands in adverse weather was challenging. On both islands we collected data over a 5-day period per month during May–July of 2015, 2016 and 2017, resulting in a total of 15 sampling occasions per island and year. Each sampling occasion consisted of five trap checks per day, with a total trapping time of 75 hours per year per island (Supplementary Table 1).

We weighed all captured individuals and measured their snout–vent length, tagged unmarked animals and recorded tags of recaptured ones. All individuals in good health (excluding gravid females, and juveniles with a snout–vent length < 64 mm or body mass < 11.5 g) were tagged with passive integrated transponders (8 × 1.4 mm, 30 mg, model ID162B FDX-B; Trovan, Douglas, Isle of Man) operating at a frequency of 134.2 kHz. Tags were inserted subcutaneously in either the left or right side of the body with a syringe implanter and 1.25-inch 14-gauge sterile disposable hypodermic needle. Prior to tagging, the injection site was wiped with antiseptic. Immediately after injection, a drop of cyanoacrylate glue was applied over the injection site to prevent tag loss and aid wound healing (Germano & Williams, 1993; Gibbons & Andrews, 2004). Once implanted, the tags were checked using a tag reader that revealed the individual's unique identification code. Even though tagging may induce temporary stress (Langkilde & Shine, 2006), if inserted properly the tags do not otherwise affect the animals (Gibbons & Andrews, 2004; Connette & Semlitsch, 2012). Once processed, we immediately released individuals at the capture site.

### Statistical analyses

We compiled capture histories as time series of binary values, with captures coded as 1 and non-captures as 0, using a standard x-matrix format (Otis et al., 1978; Nichols, 1992). Rows represented capture histories of each captured individual and columns represented capture occasions.

We based the capture–mark–recapture modelling on a robust design model (Nichols, 1992), which assumes population closure between secondary sampling occasions or days (i.e. no births, deaths, emigration or immigration during each 5-day sampling period), but assumes a population open to demographic changes between primary sampling occasions or years (over 3 years, 2015–2017; Fig. 2). The resulting encounter history consisted of 15 capture occasions with unequal time spacing; i.e. the 5 days were not always consecutive, nor was sampling undertaken on the same dates each year, because of weather conditions and logistical challenges.

### Model selection

To monitor population trends between the two skink populations, we constructed the robust design model to estimate

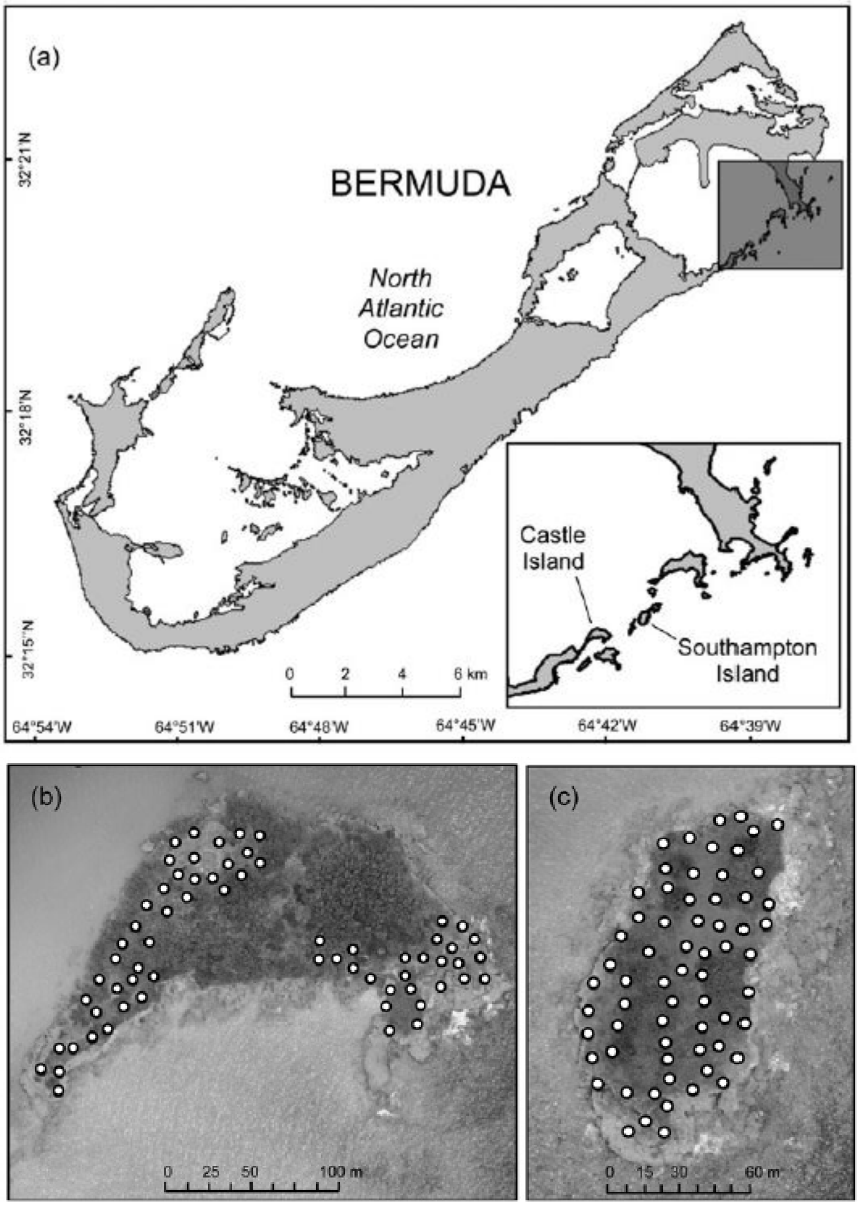


FIG. 1 (a) Bermuda with the locations of Castle and Southampton Islands within Castle Harbour, and locations of traps for the Bermuda skink *Plestiodon longirostris* on (b) Castle Island and (c) Southampton Island.

the probability of survival $\varphi$, probability of capture $p$, temporary emigration $\gamma$ and population size N, using the packages *marked*, *descr* and *Rcapture* in *R* 3.4.3 (Rivest & Baillargeon, 2014; R Core Team, 2016). We derived estimates of the demographic parameters using maximum likelihood estimates of the loglinear parameters with the *R* function *glm* and calculated standard errors by linearization.

Ten loglinear models were used to account for the time ($t$) and heterogeneity ($h$) effects of capture probabilities on the two islands. These included the model with heterogeneity effects $M_h$ (the mean probability of capture), the model with time effects $M_t$ (the capture probabilities for each capture occasion), the model with both time and heterogeneity effects $M_{th}$ (the mean probabilities of capture for each occasion) and the null model with no time or heterogeneity effects $M_o$ (the capture probability at any capture occasion; Rivest & Baillargeon, 2014). The $M_{th}$ and $M_h$ models were additionally fitted with four heterogeneity estimators: Chao (Chao, 1987), Poisson and Gamma (Rivest & Baillargeon, 2007) and Darroch (Darroch et al., 1993). In addition, we tested temporary emigration (between 2015 and 2016; and between 2016 and 2017) for each island, because the probability of capturing an individual may vary between capture occasions. The models were compared based on their Akaike information criterion (AIC) values.

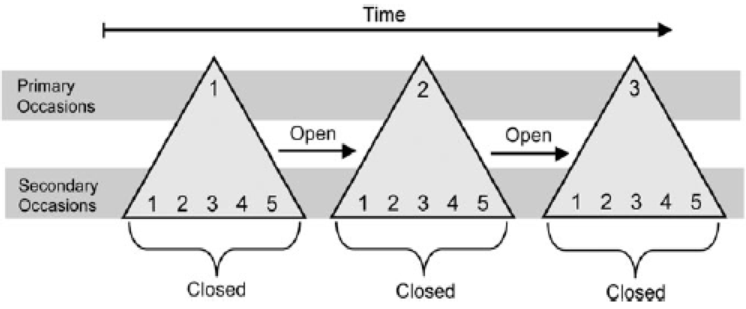


FIG. 2 Robust design example, with three primary trapping sessions each consisting of five secondary occasions.

## Results

Over 3 years (2015–2017) we captured a total of 536 skinks at the study sites. We did not tag 47 individuals caught on Castle Island (43 were too small and four escaped) and 85 individuals from Southampton Island (62 were too small, four gravid and 19 escaped). Overall, 404 individuals were tagged and we used only these in further analyses. More than half of the marked skinks were recaptured over the 3 years: 52.6% (133 of 253) on Southampton Island and 58.3% (88 of 151) on Castle Island.

Applying a robust design model, the $M_{th}$ model with Chao's estimator for each period had the lowest AIC and hence the best fit to the data for both islands (Table 1). According to the model, the estimated abundance was N = 547 ± SE 63.5 on Southampton Island and N = 277 ± SE 28.4 on Castle Island. The robust design model provided precise abundance estimates, which were calculated for each location over 3 years (Fig. 3a). Estimates appeared to fluctuate between years on both islands. When comparing mean abundance between Southampton Island (mean = 253 ± SE 105) and Castle Island (mean = 164.4 ± SE 86.9) estimates followed a normal distribution ($P < 0.05$).

Estimates of annual capture probability derived using the model $M_{th}$ were slightly higher on Southampton Island (mean P = 0.50 ± SE 0.04) compared to Castle Island (mean P = 0.42 ± SE 0.06). Capture probabilities were lowest in 2015 (Southampton Island: mean P = 0.44 ± SE 0.05; Castle Island: mean P = 0.34 ± SE 0.08) and highest in 2016 (Southampton Island: mean P = 0.57 ± SE 0.09; Castle island: mean P = 0.54 ± SE 0.08). The estimates were very precise (small standard errors) as a result of the relatively high capture probabilities (Fig. 3b).

Annual survival of skinks was higher on Southampton Island ($\varphi = 0.58$ ± SE 0.21) than on Castle Island ($\varphi = 0.40$ ± SE 0.09). Survival was lower at both sites from 2015 to 2016 but increased by 113.52% on Southampton Island and 58.72% on Castle Island from 2016 to 2017.

When comparing models, the best fit models for both islands included temporary emigration between years, indicating that a small number of individuals were not available for capture within the sampling areas. Between 2015 and 2016 temporary emigration was highest on Southampton Island but was not apparent on Castle Island. Between 2016 and 2017 temporary emigration was higher on Southampton but also occurred on Castle Island (Table 2).

## Discussion

### Population trends over time

This study provided precise estimates of abundance, capture and survival probabilities of a Critically Endangered lizard at two study sites. Our results demonstrate that passive integrated transponder tags provide a reliable method for long-term marking of lizards, and that a robust design model is effective for monitoring skink population trends.

Previous population estimates of the Bermuda skink were calculated using the simple Lincoln–Petersen method, which may be subject to bias (Seber, 1982). Using a temporary marking method (acrylic paint spots) can lead to some recaptures going undetected and the population size being overestimated as a result. In addition, previous studies did not account for juveniles (because of the trapping method used), and some were undertaken during the breeding season (May–June), with brooding females assumed to be unavailable for capture. Davenport et al. (1997) estimated the skink population on Southampton Island to be 414 in 1997, and Glasspool & Outerbridge (2004) derived an estimate of 582 in 2004. On Castle Island, Hammond (2000) estimated 116 individuals in 2000. Population estimates were also calculated in 1998 at four other sites on Bermuda; 44 individuals on Palm Island in Sandys Parish, 52 on Inner Pear Island and 123 on Charles Island (both in St. George's Parish), and 124 at Spittal Pond, a mainland nature reserve in Smith's Parish (Raine, 1998; Wingate, 1998). Although these populations showed viable proportions of juveniles and adults with breeding potential at the time, there are now concerns for their long-term survival.

Estimating survival is important in demographic studies. For example, if skink recruitment is low, then persistence, repeated breeding and longevity may be key factors for

TABLE 1 Model selection criteria for Bermuda skink *Plestiodon longirostris* capture–mark–recapture data from Southampton Island and Castle Island, Bermuda. Table shows skink abundance estimate ± SE, Akaike information criterion (AIC), difference of AIC to best-performing model (ΔAIC), degrees of freedom (df) and deviance for each model.

| Model | Estimate ± SE | AIC | ΔAIC | df | Deviance |
|---|---|---|---|---|---|
| **Southampton Island** | | | | | |
| $M_{th\ Chao}$ | 547.2 ± 63.5 | 571.53 | 0.00 | 32,740 | 332.39 |
| $M_{th\ \mathrm{Poisson}}$ | 595.5 ± 80.4 | 574.31 | 2.78 | 32,743 | 336.17 |
| $M_{th\ \mathrm{Gamma}}$ | 724.1 ± 91.6 | 574.44 | 2.91 | 32,745 | 340.30 |
| $M_{th\ \mathrm{Darroch}}$ | 719.0 ± 86.3 | 576.44 | 4.91 | 32,745 | 337.30 |
| $M_t$ | 482.2 ± 43.5 | 589.20 | 17.67 | 32,746 | 357.06 |
| $M_{h\ \mathrm{Darroch}}$ | 718.7 ± 86.2 | 607.07 | 35.54 | 32,757 | 396.93 |
| $M_{h\ \mathrm{Poisson}}$ | 601.1 ± 81.7 | 609.85 | 38.32 | 32,755 | 395.72 |
| $M_{h\ \mathrm{Gamma}}$ | 722.5 ± 91.3 | 609.86 | 38.33 | 32,757 | 399.72 |
| $M_{h\ \mathrm{Chao}}$ | 555.2 ± 65.0 | 612.34 | 40.81 | 32,752 | 392.20 |
| $M_0$ | 490.9 ± 44.7 | 622.94 | 51.41 | 32,758 | 414.80 |
| **Castle Island** | | | | | |
| $M_{th\ \mathrm{Chao}}$ | 294.8 ± 31.5 | 416.95 | 0.00 | 32,745 | 249.36 |
| $M_t$ | 274.7 ± 27.5 | 418.64 | 1.69 | 32,747 | 251.66 |
| $M_{th\ \mathrm{Gamma}}$ | 196.6 ± 41.8 | 420.45 | 3.50 | 32,743 | 247.36 |
| $M_{th\ \mathrm{Darroch}}$ | 183.6 ± 24.1 | 421.06 | 4.11 | 32,743 | 247.79 |
| $M_{th\ \mathrm{Poisson}}$ | 175.2 ± 30.7 | 421.55 | 4.60 | 32,743 | 248.27 |
| $M_0$ | 292.7 ± 30.6 | 521.41 | 104.46 | 32,759 | 380.12 |
| $M_{h\ \mathrm{Gamma}}$ | 193.8 ± 37.1 | 523.69 | 106.74 | 32,755 | 374.41 |
| $M_{h\ \mathrm{Chao}}$ | 294.8 ± 31.5 | 523.73 | 106.78 | 32,757 | 378.45 |
| $M_{h\ \mathrm{Darroch}}$ | 183.5 ± 24.1 | 523.99 | 107.04 | 32,755 | 374.71 |
| $M_{t\ \mathrm{Poisson}}$ | 176.9 ± 32.5 | 524.27 | 107.32 | 32,755 | 374.98 |

recovery. Although previous surveys state the Bermuda skink may live for up to 21 years (Davenport et al., 2001), the survival estimates from our study suggest shorter life spans, similar to the 2–7 years observed in other closely related *Plestiodon* species in the wild (Clark et al., 2005). At both sites survival was low between 2015 and 2016, which resulted in a population decrease, followed by high survival between 2016 and 2017, which led to an increase in recruitment of the larger size classes. This was most likely the result of El Niño (a cyclic shift in atmospheric patterns; Cai et al., 2017) between 2015 and 2016. Although few major impacts such as tropical storms and hurricanes affected Bermuda during that time, precipitation was increased and winds were stronger compared to 2017 (Bermuda Weather Service, 2017). These conditions would have been unfavourable for skinks, with a reduction in seasonal food abundance and fewer opportunities for foraging, basking, breeding and incubating eggs. In turn, these factors may have driven the population fluctuations on the two islands.

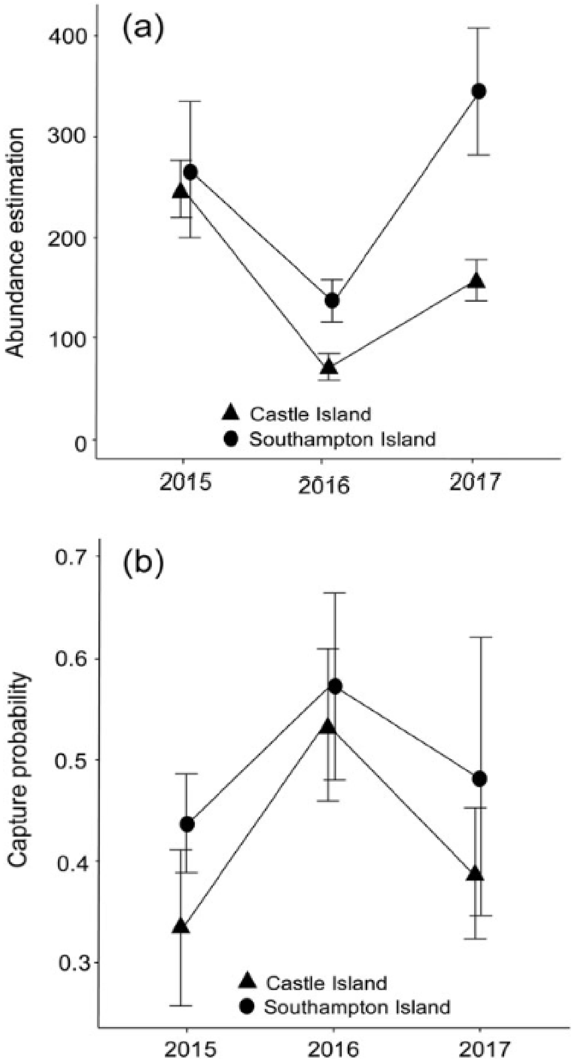


FIG. 3 (a) Comparison of $M_{th\ \mathrm{Chao}}$ robust design model of Bermuda skink abundance estimates with standard error (vertical bars) on Southampton Island and Castle Island, Bermuda, during 2015–2017. (b) Comparison of $M_{th\ \mathrm{Chao}}$ robust design model of Bermuda skink capture probability with standard error (vertical bars) on Southampton Island and Castle Island, Bermuda, during 2015–2017.

Southampton Island continues to harbour the highest density of skinks on Bermuda, mostly because landing is prohibited and threats such as introduced predators and competitors (rats *Rattus* sp., kiskadee flycatchers *Pitangus sulphuratus*, cats *Felis catus*, anolis lizards *Anolis* sp. and yellow-crowned night herons *Nyctanassa violacea*), invasive plants (asparagus fern *Asparagus densiflorus* and casuarina trees *Casuarina equisetifolia*) and anthropogenic disturbance are largely absent. However, these threats are present on Castle Island and may explain why populations there take longer to recover from decreases (Fig. 3a). Studying population dynamics over a longer period may reveal whether these estimates are cyclic or stable and may uncover factors limiting population growth.

TABLE 2 Estimated *P. longirostris* survival probability and temporary emigration, ± SE, on Southampton Island and Castle Island, Bermuda, during 2015–2017, using an $M_{th\ \mathrm{Chao}}$ robust design model.

| Sampling period | Survival probability $\varphi \pm SE$ | Temporary emigration $\gamma \pm SE$ |
|---|---|---|
| **Southampton Island** | | |
| 2015–2016 | 0.372 ± 0.08 | 39.1 ± 32.2 |
| 2016–2017 | 0.794 ± 0.19 | 36.7 ± 49.8 |
| **Castle Island** | | |
| 2015–2016 | 0.309 ± 0.06 | 0.0 ± 0.0 |
| 2016–2017 | 0.490 ± 0.19 | 26.0 ± 11.9 |

## Meeting model assumptions

Lizards may shift territories frequently during the breeding season (Ruby, 1978), or in response to fluctuations in food availability (Hews, 1993). Consequently, animals are more likely to be captured at some locations and times than others. This violates the standard assumptions of basic

Meeting model assumptions

Lizards may shift territories frequently during the breeding season (Ruby, 1978), or in response to fluctuations in food availability (Hews, 1993). Consequently, animals are more likely to be captured at some locations and times than others. This violates the standard assumptions of basic

capture–mark–recapture models (Seber, 1982; Hammond, 1986). However, the interval between the primary capture trapping sessions (c. 1 year) was sufficiently long to ensure that gains (births and immigration) and losses (deaths and emigration) would occur and the sample size collected was large enough to detect heterogeneity and for the $M_{th}$ model to fit well.

Temporary emigration may occur when a proportion of the population remains unavailable for capture. For example, skinks often spend a considerable time in rock crevices where they are difficult to capture. During the breeding season many females may be sedentary while guarding their nests (Glasspool & Outerbridge, 2004) and are therefore unavailable for capture. If temporary emigration is not tested for it can lead to negatively biased population estimates, and model assumptions may be violated (Hammond, 1986, 1990).

For the management of small, threatened populations, the potential ramifications of underestimating (negative bias) or overestimating (positive bias) abundance are clearly important. As the robust design uses two levels of sampling it allows for more parameters to be estimated and for finer control over the relative precision of each parameter (Kendall & Pollock, 1992; Kendall et al., 1995). The only major problem associated with the robust design is the large trapping effort required (Pollock, 1982); a minimum of 5 days at each site is recommended (Otis et al., 1978; Nichols, 1992) and intensive sampling can be costly. We therefore recommend this design for future capture–mark–recapture studies aimed at estimating reptile demographic parameters, particularly for species for which population declines need to be detected before they become critical.

Overall, the 10 different models in this study produced a broad range of abundance estimates, 175.2–294.8 individuals on Castle Island and 418.2–724.1 individuals on Southampton Island (both excluding juveniles). However, the abundance estimates based on $M_{th}$ are reliable and using Chao's estimator provided lower bound conservative estimates that produced better fits than other estimators such as Darroch, Poisson and Gamma, which can be highly variable (Rivest & Daigle, 2004), especially in small scale capture–mark–recapture studies (Chao, 1987). To estimate abundance with little bias, capture probability must be relatively high (Otis et al., 1978; Burnham & Overton, 1979). Skinks had a moderate probability of capture on both Southampton and Castle Island (0.33–0.54 and 0.44–0.57, respectively), and therefore the trapping method is adequate to describe the dynamics of these populations. In general, the $M_{th}$ estimator works well if most individuals are captured many times and when the population size is estimated to be > 100 individuals (Otis et al., 1978).

In 2016, population sizes were low at both sites. However, at this time capture probability was highest. Although a considerable number of skinks were caught during this time

of these populations. In general, the $M_{th}$ estimator works well if most individuals are captured many times and when the population size is estimated to be > 100 individuals (Otis et al., 1978).

In 2016, population sizes were low at both sites. However, at this time capture probability was highest. Although a considerable number of skinks were caught during this time

(243 individuals in total), 53% were recaptured individuals (compared to 12% in 2015 and 36% in 2017), which explains the lower abundance estimates.

## Conclusion

The Bermuda skink receives the highest legislative protection under the Protected Species Act (2003), and the Government of Bermuda's Department of Environment and Natural Resources is committed to undertaking conservation activities that ensure the continued survival of this unique species.

Although we identified two relatively large populations, the fluctuations we observed suggest they remain vulnerable and may not be viable in the long term without management (e.g. creation of artificial burrows for both seabirds and skinks, predator control and habitat restoration), especially during periods of low abundance or when population growth is slow. We advise the continuation of passive integrated transponder tagging as a long-term marking method and that capture–mark–recapture surveys are undertaken annually, alternating between sites to monitor population trends across Bermuda. Additionally, we recommend using robust design models for evaluating population parameters when samples are taken over multiple days and years, as these can provide timely insights into population trends and the mechanisms driving them, with important implications for future conservation and research efforts.

**Acknowledgements** We thank the Government of Bermuda Department of Environment and Natural Resources for providing the necessary permissions and for their continued support;

J. Maderios, A. Copeland, M. Meijas, N. Wellman, J. Labisko, S. Clayton-Green, D. Muldoon, N. Wright, J. Carney, D. D'Afflitto, M. Alonso, S. Massey, M. Shailer, P. Drew, R. Frith, K. Trott, R. Marirea, J.P. Rouja, P. Rouja and L. Thorne for assistance with fieldwork; R. McCrea and E. Matechou for assistance with statistical analyses; and the reviewers for their critiques. This research was funded by J. Summers Shaw, Chester Zoo, the States of Jersey, the British Herpetological Society and the Bermuda Zoological Society (Eric Clee Fund). This is Contribution #278, Bermuda Biodiversity Project (BBP), Bermuda Aquarium, Museum and Zoo, Department of Environment & Natural Resources.

**Author contributions** Analysis and writing: HT; experimental and statistical design: HT, RG; field work: HT, RG, MO, GG; revisions: RG, GG, MO.

**Conflicts of interest** None.

**Ethical standards** This research abided by the *Oryx* guidelines on ethical standards, was approved by Chester Zoo and the University of Kent Research and Ethics Committee and was conducted under permits issued by the Government of Bermuda's Department of Environment and Natural Resources (DENR 15-06-12-27). Handling and tagging of skinks were undertaken in accordance with the conditions of the licence.

## References

Bacon, J.P., Gray, J.A. & Kitson, L. (2006) Status and conservation of the reptiles and amphibians of the Bermuda islands. *Applied Herpetology*, 3, 323–344.

Bermuda Weather Service (2017) *Climate Data*. Http://www.weather.bm/climate.asp [accessed 10 May 2017].

Besbeas, P., Freeman, S.N., Morgan, B.J.T. & Catchpole, E.A. (2002) Integrating mark–recapture recovery and census data to estimate animal abundance and demographic parameters. *Biometrics*, 58, 540–547.

Burnham, K.P. & Overton, W.S. (1979) Robust estimation of population size when capture probabilities vary among animals. *Ecology*, 60, 927–936.

Cai, W., Wang, G., Santoso, A., Lin, X. & Wu, L. (2017) Definition of extreme El Niño and its impact on projected increase in extreme El Niño frequency. *Geophysical Research Letters*, 44, 11184–11190.

Chao, A. (1987) Estimating the population size for capture–recapture data with unequal catchability. *Biometrics*, 45, 427–438.

Clark, K., Hendricks, A. & Burge, D. (2005) Molecular identification and analysis of *Borrelia burgdorferi* sensu lato in lizards in the Southeastern United States. *Applied Environmental Microbiology*, 71, 2616–2625.

Connette, G.M. & Semlitsch, R.D. (2012) Successful use of a passive integrated transponder (PIT) system for below-ground detection of plethodontid salamanders. *Wildlife Research*, 39, 1–6.

Conyers, J. & Wingate, D. (1996) *Plestiodon longirostris*. In *The IUCN Red List of Threatened Species 1996*. Http://dx.doi.org/10.2305/IUCN.UK.1996.RLTS.T8218A12900393.en [accessed 11 May 2018].

Cormack, R.M. (1968) The statistics of capture–recapture methods. *Oceanography and Marine Biology Annual Review*, 6, 455–506.

Darroch, S.E., Fienberg, G., Glonek, B. & Junker, B. (1993) A three sample multiple capture–recapture approach to the census population estimation with heterogeneous catchability. *Journal of the American Statistical Association*, 88, 1137–1148.

Davenport, J., Hills, J., Glasspool, A. & Ward, J. (1997) *A Study of Populations of the Bermudian Rock Lizard (Skink),* Eumeces longirostris, *Cope (1861), on the Islands of Nonsuch and Southampton, Bermuda*. Government of Bermuda, Department of Agriculture and Fisheries, Flatts, Bermuda.

Davenport, J., Hills, J., Glasspool, A. & Ward, J. (2001) Threats to the Critically Endangered endemic Bermudian skink *Eumeces longirostris*. *Oryx*, 35, 332–339.

Edgar, P., Kitson, L., Glasspool, A. & Sarkis, S. (2010) *Recovery Plan for the Bermuda Skink* Eumeces longirostris. Government of Bermuda, Department of Conservation Services, Flatts, Bermuda.

Garber, S. (1988) *Behaviour and ecology of the Endangered endemic Bermuda rock lizard (*Eumeces longirostris*)*. Rutgers University, New Jersey, USA.

Germano, D. & Williams, D. (1993) Field evaluation of using passive integrated transponder (PIT) tags to permanently mark lizards. *Herpetological Review*, 24, 54–56.

Gibbons, J.W. & Andrews, K.M. (2004) PIT tagging: simple technology at its best. *BioScience*, 54, 447–454.

Glasspool, A. & Outerbridge, M. (2004) *A Population Re-Survey of the Bermuda Skink,* Eumeces longirostris *Cope (1861), on Southampton Island, Castle Harbour*. Bermuda Zoological Society, Flatts, Bermuda.

Hammond, P.S. (1986) Estimating the size of naturally marked whale populations using capture–recapture techniques. *Reports of the International Whaling Commission*, 8, 253–282.

Hammond, P.S. (1990) Heterogeneity in the Gulf of Maine? Estimating humpback whale population size when capture probabilities are not equal. *Reports of the International Whaling Commission*, 12, 135–139.

Hammond, M.P. (2000) *An Analysis of Bermuda Skink (*Eucemes longirostris*) Populations on Castle Island, Bermuda*. Bermuda Aquarium, Museum and Zoo, Flatts, Bermuda.

Hews, D.K. (1993) Food resources affect female distribution and male mating opportunities in the iguanian lizard *Uta palmeri*. *Animal Behaviour*, 46, 279–291.

Kendall, W.L. & Pollock, K.H. (1992) The robust design in capture–recapture studies: a review and evaluation by Monte Carlo simulation. In *Wildlife 2001: Populations* (eds D.R. McCullough & R.H. Barrett), pp. 31–43. Elsevier Science Publishers, Barking, UK.

Kendall, W.L., Pollock, K.H. & Brownie, C. (1995) A likelihood-based approach to capture–recapture estimation of demographic parameters under the robust design. *Biometrics*, 51, 293–308.

Krebs, C.J. (1999) *Ecological Methodology*. 2nd edition. Addison-Wesley Educational Publishers, Inc., Menlo Park, USA.

Langkilde, T. & Shine, R. (2006) How much stress do researchers inflict on their study animals? A case study using a scincid lizard, *Eulamprus heatwolei*. *Journal of Experimental Biology*, 209, 1035–1043.

Moore, J., Charles, A., Daugherty, H. & Nelson, N.J. (2009) Large male advantage: phenotypic and genetic correlates of territoriality in tuatara. *Journal of Herpetology*, 43, 570–578.

Nichols, J.D. (1992) Capture–recapture models. *Bioscience*, 42, 94–102.

Otis, D.L., Burnham, K.P., White, G.C. & Anderson, D.R. (1978) Statistical inference from capture data on closed animal populations. *Wildlife Monograph*, 62, 1–135.

Pollock, K.H. (1982) A capture–recapture design robust to unequal probability of capture. *Journal of Wildlife Management*, 46, 757–760.

R Core Team (2016) *R: a Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria.

Raine, A. (1998) *A study of the morphological differentiation, fluctuating asymmetry and the threats facing isolated*

**NATURAL HISTORY NOTES 199**

*PLESTIODON LONGIROSTRIS* (Bermuda Skink). **TAIL BIFURCATION.** *Plestiodon longirostris* is the only endemic terrestrial vertebrate in Bermuda. It typically occurs along the rocky coastline, associated with the native Sea Oxeye (*Borrichia arborescens*) and Bay Grape (*Coccoloba uvifera*) vegetation (Edgar et al 2010. Bermuda Skink Recovery Plan. Department of Conservation Services, Government of Bermuda. Bermuda). However, populations remain fragmented and isolated throughout the mainland and offshore islands due to habitat loss, anthropogenic disturbances (particularly from coastal developments) and increased predation pressure and competition from several introduced species. These include rats (*Rattus rattus and R. norvegicus*), Kiskadee Flycatchers *(Pitangus sulphuratus)*, Yellow-crowned Night Herons *(Nyctanassa violacea)*, domestic and feral cats (*Felis catus*), and three *Anolis* species (*A. grahami, A. leachi, A. extremus*). While conducting fieldwork in Bermuda on *Plestiodon longirostris* during 2015–2016, we recorded the incidence of bifurcated tails in two populations.

Seven adult *P. longirostris* with bifurcated tails were discovered, the first such cases officially documented in this critically endangered species. Individuals were caught from two offshore islands within Castle Harbour; Castle Island (32.3408°N, 64.6722°W), a 3.5-acre nature reserve, and Southampton Island (32.3422°N, 64.6675°W), a 2.2-acre nature reserve. Overall (in both years), the incidence of bifurcation was 0.8% (2/238 skinks captured) on Castle Island and 1.9% on Southampton Island (5/268). The bifurcated tails differed from the original tails in coloration and scalation patterns. Two individuals were recaptured once each in May 2016 and one was recaptured three times between May and June 2016. Skinks were found to have particularly high mutilation rates: 49% of individuals captured during the study exhibited digit loss, tail loss, body scars or a combination of these.

Caudal autotomy is likely to be disadvantageous, making the lizard more conspicuous to predators and potentially reducing activity, modifying foraging behaviour and reducing mating opportunities (Chapple and Swain 2002. Funct. Ecol. 16:817–825). However, a two-sample t-test showed no difference in body mass or SVL between skinks with bifurcated tails and skinks with normal tails (N = 541, BM: 19.33 ± 4.3 g, range 11–30 g, t = 0.16, d.f. = 546, P = 0.874; SVL: 81.99 ± 5.2 mm, range 60–94 mm, t = 0.15, d.f. = 546, P = 0.881).

Observations of bifurcated tails among reptile taxa are not uncommon, have previously been reported in other scincid lizards, and are apparently not due to contaminants or genetic factors (Brindley 1898. J. Bombay Nat. Hist. Soc. 11:680–689). We hypothesize that the seven individuals we encountered survived predator attacks, which caused a large enough wound on the tail for physiological cues to trigger regeneration of an additional tail tip at the site of injury, resulting in two tails (Koleska and Jablonski 2015. Ecol. Mont. 3:26–28).

Fig. 1. Dorsal (A) and ventral (B) views of a *Plestiodon longirostris* with a bifurcated tail.

We thank the Government of Bermuda's Department of Environment and Natural Resources for protected species permits and for access to sites, and we thank everyone that assisted in the field. This work was supported by Chester Zoo, UK and the Bermuda Zoological Society. This is Contribution #252, Bermuda Biodiversity Project (BBP), Bermuda Aquarium, Natural History Museum and Zoo, Department of Environment and Natural Resources.

**HELÉNA TURNER** (e-mail: hturnerjsy@hotmail.com) and **RICHARD A. GRIFFITHS**, Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Marlowe Building, Canterbury, Kent, CT2 7NR, UK; **GERARDO GARCIA**, Chester Zoo, Caughall Road, Upton by Chester, Chester, CH2 1LH, UK; **MARK E. OUTERBRIDGE**, Department of Environment and Natural Resources, P.O Box FL 145 FL BX, Bermuda.

*Herpetological Review 48(1), 2017*

m elev. 1987. Eric R. Pianka. Verified by Carol Spencer. Museum of Vertebrate Zoology, University of California at Berkeley (MVZ 214876). Confirmation of a county record with a voucher specimen. An earlier record for Blanco County reported by Raun and Gehlbach (1972. Dallas Mus. Nat. Hist. Bull. 2:1–61) and cited by Dixon (2013. Amphibians and Reptiles of Texas: with Keys, Taxonomic Synopses, Bibliography, and Distribution Maps. Texas A&M University Press, College Station, Texas. 447 pp.) included no locality information or voucher specimen number and therefore cannot be verified. One individual was taken on rolling ranchland by a house cat and taken from the cat by Pianka, who gave it to Kevin de Queiroz, who deposited the specimen in the Museum of Vertebrate Zoology.

**RALPH W. AXTELL**†, Department of Biological Sciences, Southern Illinois University, Edwardsville, Illinois 62026, USA; **KEVIN DE QUEIROZ**, Division of Amphibians and Reptiles, National Museum of Natural History, Smithsonian Institution, Washington, DC 20560 USA (e-mail: dequeirozk@si.edu). † Deceased.

***PHELSUMA GRANDIS*** (Madagascan Day Gecko). USA: FLORIDA: Miami-Dade Co.: private residence in Miami (25.71668°N, 80.31043°W; WGS 84). 5 June 2017. C. Thawley. Verified by Kenneth L. Krysko. Florida Museum of Natural History (UF 179934; photo voucher). New county record (Krysko et al. 2011. Atlas of Amphibians and Reptiles in Florida. Final report, Florida Fish and Wildlife Conservation Commission, Tallahassee, Florida. 524 pp.). Further surveys at this locality (15 June 2017 and 6 July 2017) have documented numerous mature individuals of both sexes as well as hatchlings, suggesting the presence of a breeding

***PLESTIODON LONGIROSTRIS*** (Bermuda Skink). BERMUDA: ST. GEORGES: Castle Harbour: North Cock Rock (32.34589°N, 64.66633°W; WGS 84) and adjacent South Cock Rock (32.34494°N, 64.66659°W; WGS 84), 0.8 m elev. 10 May 2016. Heléna Turner. Verified by Jeremy L. Maderios and Mark E. Outerbridge. National Museum of Natural History, Smithsonian Institution (USNM 2898–2900; photo vouchers). First records for the two islets (ca. 0.4 acres each) from within Castle Harbour for this critically endangered lizard (Conyers and Wingate 1996. The IUCN Red List of Threatened Species, e.T8218A12900393). Two adult skinks were caught in pitfall traps on North Cock Rock and one other adult was caught on South Cock Rock as part of an island-wide *P. longirostris* monitoring survey carried out during 2015 through 2017 across Bermuda under the Government of Bermuda's Protected Species license; all were released on site. All three lizards appeared to be in good health. The closest skink population to these two islets is on Nonsuch Island, a protected nature reserve ca. 100 m to the northeast.

**HELÉNA TURNER**, Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Marlowe Building, Canterbury, Kent, United Kingdom, CT2 7NR, UK; e-mail: ht247@kent.ac.uk.

### SQUAMATA — SNAKES

***ARIZONA ELEGANS ELEGANS*** (Kansas Glossy Snake). USA: TEXAS: Palo Pinto Co.: ca. 25.9 air km SE of Graham, Texas, near the Possum Kingdom Airport (32.91813°N, 98.43719°W; WGS

*Herpetological Review 48(4), 2017*

June 2018, Issue 218

# Bermuda Skink Project

**The BHS student grant was awarded to Heléna Turner in 2017 to assist with field equipment, particularly PIT tags needed for her Ph. D research on the critically endangered Bermuda skinks (*Plestiodon longirostris*).**

This study has been assessing the distribution, abundance and status of the Bermuda skinks. It represents the most comprehensive review and long-term surveying of this species to date. Once common throughout Bermuda, skink populations have been declining since 1965 due to several factors including habitat loss and fragmentation, anthropogenic activities and the introduction of multiple invasive species (Glasspool & Outerbridge 2005).

Figure 1. Adult Bermuda Skink (Plestiodon longirostris).

Figure 2. Skinks caught in pit fall trap (under licence from the Government of Bermuda).

Using one-gallon glass jars as traps and a small amount of cheese or rotten fish as bait, island wide surveys were undertaken between 2015-2017. Skinks were present at 12 sites (31.6%) of those surveyed and 95.6% were found in the east of the Island within Castle Harbour, confirming that in the event of a hurricane the population is extremely vulnerable.

A total of 253 individual skinks were marked with PIT (Passive Integrated Transponder) tags that were captured and subsequently recaptured 1078 times. The tags are not on-

Figure 3. Taking various morphometric measurements.

9

June 2018, Issue 216

ly useful for short term on the mark-capture-recapture aspect of the study but will be beneficial for the long term by providing an insight into the population dynamics such as growth and survival rates for the first time with this species.

Figure 4. Bermuda Skink Hatchling caught on Castle Island.

By evaluating the efficiency of a capture–mark–recapture population estimation method and robust-design model we are able to monitor trends in abundance, density, survival and capture probabilities. We found the robust-design model provided precise estimates of abundance (N±SE) at the two largest Bermuda skink sub-populations were 258 ± 4.5 on Southampton Island and 157 ± 4.4 on Castle Island.

Overall, we found the populations did not appear to be stable and fluctuated at both sites over the 3-year period. As a result, we found that reliable trapping estimates can provide accurate early-warning signals in advance of the decline of the breeding population, so action can be taken in time to ensure populations remain stable.

Figure 5. Pit Tagging a skink on Nonsuch Island.

As a result of island wide surveys, skinks were also found on North Cock Rock and South Cock Rock for the first time (Turner. 2017) and for the first time we observed 2% of individuals had bifurcated tails that were captured on island populations most likely the result of increased predation (Turner *et al.* 2017).

10

Figure 7. Bifurcated Tail observed in Southampton Island Skink.

Our results emphasize the importance of estimating reliable population parameters that can provide timely insights into population trends and the mechanisms driving them, which has important implications for the future conservation and research effort, to help prevent extinction.

Further analysis is currently being conducted to see if sub-populations are morphologically or genetically distinct. The data collected from this study will be vital to be able to assess the current size and status of the population that will be used to inform conservation management and used in the implementation of future monitoring programmes.

Figure 6. Reading the individuals unique PIT tag number.

## References

Glasspool, A., and Outerbridge, M. (2005). A population resurvey of the Bermuda skink, Eumeces longirostris cope (1861), on Southampton Island. (Bermuda Zoological Society: Bermuda).

Turner, H., Griffiths, R., Garcia, G., & Outerbridge, M. (2017). Natural History Notes: Plestiodon Longirostris Bermuda Skink Tail Bifurcation. Herpetological Review, 48(1), 198-199.

Turner, H (2017). Geographic Distribution Notes: Plestiodon Longirostris Bermuda Skink. Herpetological Review, 48(4), 812.

***PLESTIODON LONGIROSTRIS*** (Bermuda skink). **ECTOPARASITISM*.*** During a mark-capture-recapture study on the Bermuda skink between May – July 2016, several individuals were found to have ectoparasitic mites present, located inside the tympanic membrane. This finding represents a new host and locality record and what is the first ectoparasitic mite (Acari) species observed in Bermuda skinks *Plestiodon longirostris* (Sauria: Scincidae).

Surveys were conducted throughout Bermuda's mainland and offshore islands but only individuals on Southampton Island (32.3422°N, -64.6675°W), a 2.2-acre nature reserve and Castle Island (32.3408°N, -64.6722°W), a 3.5-acre nature reserve, Castle Harbour (the two largest remaining sub-populations of Bermuda skinks) had mites present.

Thirty adult skinks [adult snout to vent length (SVL): 72-94 mm and 12-26 g body mass] were found to have mites, 11.42% from Southampton Island (24/274) and 15.17% from Castle Island (6/91).

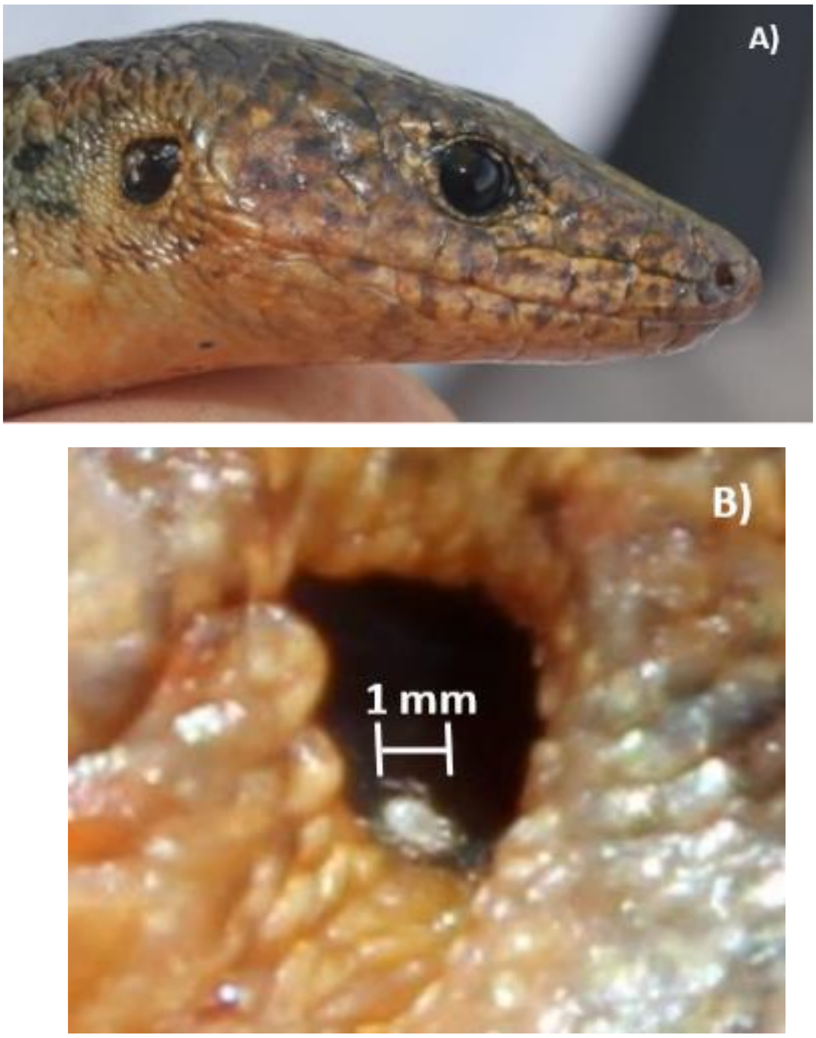
Twenty-one skinks had mites in just one side of their tympanum and the remaining nine had the mites in both sides with on average 3.6 mites per lizard with a maximum of thirteen mites. The adult skinks tympanum is between 5 – 6 mm in diameter and the mites were ≤ 0.05 mm in comparison (recorded using digital callipers: precision ± 0.01 mm; Fig.1).

The high densities of lizards on these islands may increase host susceptibility to parasites. No juveniles or sub-adults or any individuals from other locations were noticed to have mites present. Ectoparasites are generally seasonal in abundance so it is possible they are present in other populations but were not observed. Although the effect of mites on host fitness is unclear it is hypothesised to be parasitic relationship and that the mites most likely arrived to the islands via invasive bird, rat or anole species. Although all skinks found with ear mites appeared to be in optimum health, if they are found to be living off the skinks blood supply and/or breeding within the tympanum it could potentially lead to bacterial infection transmission and alternations in blood composition leading to abnormal behaviour, have an impact on auditory cues or reduced survivorship as a result and should be further investigated in this critically endangered lizard.

We thank the Government of Bermuda Department of Environment and Natural resources for protected species permits and access to sites, with special thanks to M.E. Outerbridge, R.A. Griffiths, G. Garcia and J. Maderios for their support.

**HELÉNA TURNER**, Durrell Institute of Conservation and Ecology, School of Anthropology and Conservation, University of Kent, Marlowe Building, Canterbury, Kent, United Kingdom, CT2 7NR, UK, email: ht247@kent.ac.uk.

FIG. 1. (A) An adult skink with several (Acari sp.) mites within the right tympanum: only observed in Southampton Island and Castle Island sub-populations (B) Close up of a mite within a skink's tympanic membrane.



# SUPPLEMENTARY MATERIAL 6. – Supporting Media

## ARTICLES

- CHESTER ZOO SCIENCE REVIEW 2015
- CHESTER ZOO SCIENCE REVIEW 2017

Chester Zoo Conservation Scholars

**Project: Population status and conservation of the critically endangered Bermuda skink *Plestiodon longirostris*, Heléna Turner, University of Kent.**

*'I am researching the population and conservation of a critically endangered lizard, the Bermuda skink. As they have many threats, I want to know how many currently remain in the wild and determine whether there have been any significant changes in the health of the population to help with their future recovery.'*

# BIODIVERSITY SURVEYS AND ECOLOGICAL MONITORING

**Understanding basic ecological parameters is essential in order to implement effective conservation. R**
**and accurate surveys are key research tools required to implement carefully designed monitoring initiative**
**staff and partners are experts at surveying a wide variety of biodiversity in all of our regional programmes**

## Losing lizards in the Bermuda Triangle: Assessing the status of the Critically Endangered Bermuda skink

*By Heléna Turner, Conservation Scholar*

**Heléna Turner, Conservation Scholar and PhD student from the University of Kent, tells us more about her research on the population and conservation of the critically endangered Bermuda skinks.**

A wide variety of sampling methods have been used in monitoring studies such as capture mark-recapture techniques to estimate abundance, capture probability and survival probability of wildlife populations (Krebs 1999). Abundance and survival are critical determinants of a population's viability (White & Burnham 1999) and important for implementing effective conservation strategies. However, the choice of sampling methods can influence the reliability of reptile surveys, which is an important consideration when unbiased estimates for secretive, nocturnal or infrequently active species are required (Beebee 2013).

Cheese and rotten fish were used as bait to attract the Bermuda skinks

Although some trapping and monitoring techniques can have relatively high capture rates, they may be seasonally biased or influenced by dietary and habitat preferences, behavioural attributes and body size of the target species (Crosswhite et al.
1999). During my PhD I have been assessing the distr
abundance and status of the critically endangered Be
skink (*Plestiodon longirostris*). This study represents th
comprehensive review and long-term survey of this spe
date. Once common throughout Bermuda, skink popu
have been declining since 1965 due to several factors in
habitat loss and fragmentation, anthropogenic activiti
the introduction of multiple invasive species (Glasspo
Outerbridge 2005).

Bermuda Skink (*Plestiodon longirostris*)

Using one-gallon glass jars as traps and a small amount of cheese or rotten fish as bait, the research team undertook island-wide surveys between 2015 and 2017 across a grid of traps shown in figures 1 and 2. We found skinks in 12 sites which represents 31.6% of all sites surveyed. A total of 253 individual skinks were marked with PIT (Passive Integrated Transponder) tags and were captured and subsequently recaptured 1,078 times.

By evaluating the efficiency of a capture–mark–recapture population estimation method and robust-design model, we were able to monitor trends in abundance, density, survival and capture probabilities, of the two largest Bermuda skink sub-populations in Bermuda, Castle Island and Southampton Island. We found that the robust-design model provided precise estimates of abundance where we estimated an abundance of 258 skinks (± SE 4.5, CI 255–275) on Southampton Island and an estimated abundance of 157 skinks (± SE 4.4, CI 153–172) on Castle Island. We estimated the density to be of 318 skinks/Ha on Southampton Island and 111 skinks/Ha on Castle Island. We also showed that capture probabilities were highest on Southampton Island (mean

$\dot{\bar{P}}$ = 0.50 ± 0.09) compared to Castle Island (mean, $\dot{\bar{P}}$ = 0.42 ± 0.07).

Although the two islands are roughly 400 metres apart and are similar in abiotic environmental conditions the main differences between the two are largely due to two main factors. 1) Caste Island is located on the cusp of the mainland and continues to be a popular location for locals and tourists to anchor boats and to swim and jump off the island, disturbing coastal habitat and leaving litter. In comparison, Southampton Island is much more difficult to access and is not affected by anthropogenic disturbances. 2) In order to try and prevent people from landing on Castle Island the vegetation is minimally managed. This in turn creates dense coastal forest habitat that is perfect for invasive anoles, kiskadees, herons and rats, all of the skinks main predators so although Castle Island is much larger the population is substantially lower due to these continued threats.

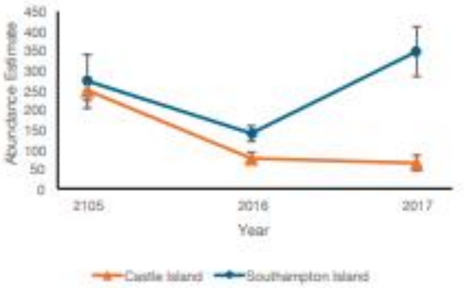
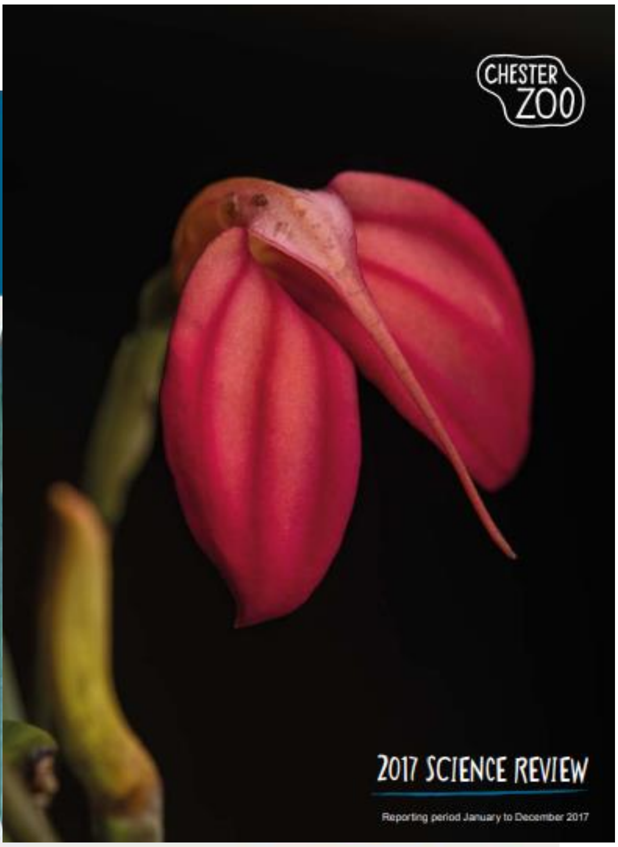


**Figure 1.** Comparison of robust design model abundance estimates with standard error ($N_i$ ± SE) of Bermuda skinks Plestiodon longirostris on Southampton Island and Castle Island, Bermuda.

Overall, we found that the populations did not appear to be stable and fluctuated at both sites over the three-year period (see figure 1). As a result, we found that reliable trapping estimates can provide accurate early-warning signals of the decline in the breeding population so action can be taken in time to ensure populations remain stable.

Further analysis is currently being conducted to see if sub-populations are morphologically or genetically distinct. The data collected from this study will be vital to assess the current size and status of the population that will then be used to inform conservation management and used in the implementation of future monitoring programmes.



**References**

Beebee, T. J. *Amphibians and reptiles* (Pelagic Publishing, 2013).

Crosswhite, D. L., Fox, S. F., Thill. R. E. *Oklahoma Academy of Science Proceedings* **79**, 45–50 (1999).

Glasspool, A., Outerbridge, M. *A population re-survey of the Bermuda skink, Eumeces longirostris cope* (1861), on Southampton Island (Bermuda Zoological Society, 2005).

Krebs, C. J. *Ecological Methodology, 2nd ed.* (Addison-Wesley Educational Publishers, Inc, 1999).

White, G. C., Burnham, K. P. *Bird Study* **46**, 120-139 (1999).

## Key findings

**96%** of skinks were found in the east of the island within Castle Harbour confirming that in the event of a hurricane the population is extremely vulnerable.

**258** Bermuda skinks were estimated to be on Southampton Island and 157 on Castle Island.

Estimating reliable population parameters can provide timely insights into population trends and the mechanisms driving them and has important implications for the future conservation and research effort to help prevent extinction.

# SUPPLEMENTARY MATERIAL 7. – Conference report

## PRESENTATIONS

- Turner, H. 22nd February 2018. Losing lizards in the Bermuda Triangle. Chester Zoo Science Seminar.
- Turner, H. 18th March 2017. Losing lizards in the Bermuda Triangle. The British Herpetological Society (BHS). Wildwood, Herne Bay, Kent.
- Turner, H. 15th January 2017. Losing lizards in the Bermuda Triangle. Kent Amphibian and Reptile Group (KRAG) AGM Meeting. Tyland Barn, Kent Wildlife Trust, Kent.

**Losing Lizards in the Bermuda Triangle:**

**Conserving the critically endangered Bermuda skinks**

**Heléna Turner**
PhD Candidate
Durrell Institute of Conservation and Ecology
University of Kent

DICE
University of Kent

December 2016 The NatterJack

The British Herpetological Society

# 70th Annual General Meeting
# Saturday 18th March 2017

Wildwood Education Centre
Herne Common, Herne Bay
Kent, CT6 7LQ

The British Herpetological Society

wildwood

## AGENDA

12.00 Registration, refreshments (provided)

12:30 Introductions (Chair: Chris Gleed-Owen)

12:35 **Rick Hodges** (KRAG):
Getting to know adders personally: case studies on adder behaviour.

13:05 **AGM Business Meeting**, including: Approval of 2016 AGM Minutes, Matters Arising, Chairman's Report, Treasurer's Report, Council for 2017-18, 2017 Subscription fees
AOB

13:40 Break for Lunch

14.40 **Helena Turner** (DICE):
Losing Skinks in the Bermuda triangle

15:15 Refreshments (provided)
Photographic Competition judging

16:30 Close

KRAG

DICE
University of Kent

Issue 213 Page 2

## POSTER PRESENTATION

- Turner, H (2017) Losing lizards in the Bermuda Triangle. Poster Presentation Herpetofauna Workers Meeting (ARC UK), Oxford.

30 November 2015

The hugely popular Herpetofauna Workers Meeting was held in Oxford in 2016, with two days of fascinating talks and workshops, the latest news on herpetofauna conservation and plenty of networking opportunities. As ever, the parallel social programme also proved very popular, with many attending the ARGUK curry on Friday night, and the Saturday Gala Dinner with the perennial favourite HIGNFY, which this year featured herpy super heroes, and a new Olympic sport of 'toad tossing' (plastic ones I hasten to add!).

For a more personal summary, I include a review from our colleagues at KRAG:

The 2016 Herpetofauna workers meeting included a range of talks from local volunteers, students and professionals. For me, some of the stand out presentations were those that illustrated community participation and talks that summarised the results of monitoring projects. Amy Wright's talk on Kent Toad Crossings being an excellent example of how a dedicated volunteer can motivate people to get involved in conservation. John Baker's account of adder monitoring in East Anglia demonstrated the importance of long-term data sets, inspiring a discussion in the bar that concluded with plans for a further autumn conference and a commitment to publish evidence that can be used to influence land managers involved in practical conservation.

The meeting also allowed students to present results of their ongoing research projects. Aidan Mackay's talk on Kent marsh frogs and Rob Ward's account of Jersey grass snakes were both delivered confidently with interesting findings. I also enjoyed learning about Layna's island skinks in Helena Turner's nicely presented poster. Posters are a great way of summarising projects and perhaps something to be expanded for future events.

Finally, the workshops provided a platform for statutory organisations to announce new publications and planned changes to the interpretation of conservation legislation.