Assessing the Impacts of Artificial Light and Anthropogenic Noise on Seabirds

By

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ABSTRACT

Seabirds are important bio-indicators within marine environments and are commonly used by researchers to monitor pollutant levels and inform conservation efforts as they are long-lived species that are highly reliant on their senses and exploit a board range of habitats and therefore the occurrence of rapid changes in seabird colony dynamics can indicate the presence of extremely harmful environmental conditions. Coastal developments are ever expanding and increasing the presence of harmful pollutants within the marine environment. Artificial light and anthropogenic noise are common sensory pollutants produced from coastal developments yet their impacts on seabirds are not fully studied. In this thesis our aim was to determine the behavioural and physiological impact anthropogenic noise, such as construction noise, on incubating little penguins, while also monitoring any impacts to breeding success of little penguins and of closely associated seabird species, the black-faced cormorant and the crested tern. We used cameras to record behavioural response and a dummy egg contained with an internal omnidirectional lavalier condenser microphone connected to a either a Zoom H4n or a Tascam DR-05 to record the heart rate of the individual. We also aimed to determine the behavioural impacts artificial light would have on returning little penguins at night. For this experiment we used a self-sustaining light system, composed of one white LED floodlight connected to an inverter linked to a deep cycle battery that was automatically charged by one Solar Panel and had a control system with a decoy light that mimicked the experimental system at two different landing sites. This thesis found a significant increase in vigilance behaviours of little penguins exposed to construction noise, and a significant increase in duration of behavioural recovery from construction playback, with this short-term experiment having no negative impact on their breeding success. We also found that under artificially illuminated conditions little penguins tended to arrive later on nights when the light was on compared to night when the light was off and that occurrence of vigilance behaviours was site-specific and significantly less when the light was on. Our results reveal sensory disturbance, such as artificial light and anthropogenic noise, produced from coastal developments can initiated cause individuals to alter their behavioural time budget.

Keywords: Light, Noise, Seabirds, Little penguin, Behavioural, Physiological, Impacts

DECLARATION

I certify that this thesis does not incorporate without acknowledgment any material previously submitted for a degree or diploma in any university; and that to the best of my knowledge and belief it does not contain any material previously published or written by another person except where due reference is made in the text.

Signed..

Date.......27th May 2022...................

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CHAPTER 1 – SENSORY DISTURBANCE

1.1 Human disturbance and its impact on wildlife's sensory environment

The advancement of human developments has led to the increased output of harmful pollutants into the environment that create altered conditions for wildlife to live in (Boyle & Samson 1985; Wiese et al. 2001; Fujii et al. 2016; Rodríguez et al. 2015). Coastal habitats have become popular settlement sites for humans (Small & Nicholls 2003; Balk et al. 2009; Nations 2017) as they provide numerous services, from raw resources (e.g., food, water, raw materials) to cultural services (e.g., aesthetic values, tourism, cognitive effects), all of which leading to a growing use of pollutants (Huppatz 2005; Xie et al. 2013; Liquete et al. 2013; Akber et al. 2020). Such pollutants can disrupt the sensory environment of the surrounding wildlife (Boyle & Samson 1985; Wiese et al. 2001; Fujii et al. 2016; Rodríguez et al. 2015), as all animals rely on their senses for communication, orientation, foraging, breeding, and to avoid danger (Boyle & Samson 1985; Tablado & Jenni 2015; Blackwell et al. 2016; Blumstein 2016). The introduction of a stimulus produced from human developments and their pollutants can cause animals to become scattered and unfocused (Laughlin 2001), as well as significantly reduce an individuals' fitness (Calow & Forbes 1998; Chan & Blumstein 2011; Siemers & Schaub 2011; Thushari & Senevirathna 2020) and alter population dynamics (Lee 1998; Patisaul & Adewale 2009; MacDougall et al., 2013). The two most common stimuli produced from human developments are anthropogenic noise and artificial light, yet their impacts on wildlife through sensory disturbance can be mixed (e.g., Lowry et al. 2011; Rodríguez et al. 2017, 2018; Ditmer 2020), highlighting the needs for further research across taxa.

Human developments are a common source of artificial light due to the benefits for humans, such as increased visibility and mobility at night, but it can also disrupt the natural day-night routines of wildlife resulting in more diurnal activity at night (Russart & Nelson 2018; Hasselt 2021). As such, artificial lights can change the behavioural time budgets of individuals and alter their circadian rhythm, which may be beneficial for some species or expose them to a wide array of new threats. For example, Rodríguez et al. (2015) found that nocturnal petrel species (*Procellariiformes*) were attracted to artificially illuminated areas, detouring them from participating in foraging activities, causing grounding events and reducing survival rates. Petrels found in these illuminated areas became confused and disorientated triggering them to fly into structures or the ground and resulting in injuries or mortality (Rodríguez et al. 2015). Contrary to this, Rodríguez et al. (2018) found that little penguins (*Eudyptula minor*) returning to their nests at night preferred artificially illuminated paths as it increased their visibility and allowed for effective navigation through their environment and increased ability to detect approaching predators. Diurnal species may also use this increased visibility at night to spend more time foraging, as found by Ditmer et al. (2020) on mule deer (*Odocoileus hemionus*) and cougars (*Puma concolor*). In an ever-expanding urbanised world, there is therefore an urgent need to investigate further how artificial light at night will impact the surrounding wildlife to determine species most at risk and to inform protection zones or mitigation strategies.

Noise pollution is another common pollutant associated with human activity (Kunc & Schmidt 2019) that can originate from a variety of sources (e.g., people's voices, vehicles, boats, aircrafts, and machinery) and cause an altered attention span in animals (Sordello et al. 2020; Communication 2020). Animals in the wild have adapted to the 'normal' ambient sounds of nature (such as wind, rain, native species calls), and are expected to know how to response to these 'normal' sounds (Chan & Blumstein 2011). Depending on how developed an animal's hearing is and its life history, its reaction to unfamiliar sound can vary from an intense state of stress with long-term impacts on their fitness (Sierra-Flores et al. 2015) to an initial reaction followed by a quick return to their daily tasks (Lowry et al. 2011). For example, Lowry et al. (2011), found that noisy miners (*Manorina melanocephala*) living in urban environments became bolder when they heard an unexpected sound, while those living in rural environments completely fled the area (Lowery et al. 2011). Introduced noises can also distract an animal and reduce their efficiency or time spent performing biological important tasks (Chan & Blumstein 2011). For example, Siemers and Schaub (2011) find that introduced traffic noise within the foraging areas of the greater mouse-eared bat (*Myotis myotis*) led to reduce hunting efficiency by masking their echo calls used for hunting, thereby reducing their fitness. Researchers have also found that consistent noise levels above 70 dB can cause damage or completed loss of hearing in humans as well a wildlife (Government 2015; Breitzler et al. 2020). As machinery and heavy traffic can produce noise levels between 70 – 130 dB (Siemers & Schaub 2011; Breitzler et al. 2020; Communication 2020), the introduction of continuous anthropogenic noises generated from developments has the potential to damage wildlife hearing capabilities as well as their behaviours and ultimately to reduce their fitness.

1.2 Individual response to sensory disturbance

1.2.1 Behavioural response to sensory disturbance

A disruptive stimulus can invoke an energy-demanding "anti-predator response" from an individual where long-term exposure can lead to fatigue, suppressed immune system, increase vulnerability to predation, and reduce breeding success (Tablado & Jenni 2015; Blackwell et al. 2016; Blumstein 2016; Buxton et al. 2017a; Boyle & Samson 1985). An "anti-predator response" occurs when an individual becomes highly cautious and focuses their brain function on quick reaction and avoidance behaviours, which indicates that the individual has traversed into a stressed state (Derose-Wilson et al. 2015; Franks 2017; Bevan et al. 2018). For example, Checker-ed puffers (*Sphoeroides testudineus*) inflate their bodies to deter predators but become fatigued and unable to maintain the anti-predator response when repeatedly exposed to stressors (Cull et al. 2015). The forced over-use of energy-demanding behaviours, such as anti-predator responses, can leave individuals weak and unable to defend themselves from unexpected threats and thereby reduce survival. A similar problem may arise when the individual is undergoing an already stressful period in its life cycle, such as breeding. For example, double-crested cormorants (*Phalacrocorax auratus*) disturbed by human presence invoked an avoidance response where they abandoned their nest and left their egg(s)/chick(s) exposed to predation by gulls or ravens (Ellison & Cleary 1978). This not only deletes the energic investment of breeding adults but also reduces the number of fledglings produced in that breeding season. Therefore, unnecessary disturbance of animals can lead to drainage of energy supplies and possibly direct harm to the individual or surrounding individuals.

Another provoked response from an individual in light of a new stimulus is “the distracted prey hypothesis”, as proposed by Chan and Blumstein (2011). According to this hypothesis foreign and continuous stimuli can distract animals from preforming biologically important tasks, such as predator detection, foraging or sleeping (Chan & Blumstein 2011). For example, Chan et al., (2010) found that Caribbean hermit crabs (*Coenobita clypeatus*) distracted by boat motor playbacks had a delayed response to simulated predators before invoking an “anti-predator response”, with the noise either masking the predator’s movement or due to the noise holding the crab’s attention and distracting them from the approaching predator. In another example, Fanning et al. (2020) monitored captive Fiordland penguins (*Eudyptes pachyrhynchus*) during days where their host zoo had concerts. They found that individuals became stressed (e.g., displayed more vigilance behaviours), restless (e.g., increased their movements), and dis-harmonized (e.g., greater variation in behaviours between individuals) (Fanning et al. 2020). It should be noted however that interruption to an animal’s daily routine can occur naturally due to temporal factors (e.g., time of year, time of day, breeding season, hibernation period, moulting period; Challet et al. 1994), metabolic demands (e.g., due to high energy demand food source distribution/behaviour; Nielsen & Watt 1998), or as a result of evolutionary history (i.e., how an individual has adapted to live in a certain environment; Laughlin 2001). Yet when human activity disrupts the sensory environment, individuals may increase the time they spent performing energy-demanding behaviours, such as vigilance behaviours, instead of preserving their energy for preservation behaviours, such as foraging (Franks 2017), self-care (Fanning et al. 2020), off-spring care or anti-predator behaviours (Boyle & Samson 1985; Derose-Wilson et al. 2015; Buxton et al. 2018). The distraction and reallocation of an individual’s attention can therefore lead to reduced quality of health, due to a lack of time spent undergoing preservation activities, and reduced survival due to delayed predators’ evasion.

1.2.2 Physiological response to sensory disturbance

Monitoring the internal and physiological reactions that can inhibit an individual’s performance during essential lifecycle periods is essential to investigate how much a disturbed animal is stressed by an introduced stimulus (Nimon et al. 1995; Ellenberg et al. 2006; Reeder et al. 2012). Individuals in a stressed stated can exhibit elevated heart rate and altered production of hormones, which can disrupt homeostasis, the self-regulation of internal processes (Nimon et al. 1995). These physiological reactions can consume large amounts of energy, wasting precious fat reserves that individuals build up in anticipation for important lifecycle periods, such as hibernation, breeding or moulting (Tablado & Jenni 2015; Blackwell et al. 2016; Blumstein 2016). For example, Humboldt penguins (*Spheniscus humboldti*) doubled their heart rate when approached by humans and need at least 30 minutes to recover, further exacerbating energy expenditure initiated by the disturbance. Stressed breeding individuals either abandoned their nests or were unable to forage optimally, due to wasted energy reserves, leading to reduced food provisioning for chicks (Ellenberg et al. 2006). French et al., (2010) found that breeding marine iguanas (*Amblyrhynchus cristatus*) exposed to tourists lead to stress-induced elevations in plasma corticosterone which suppressed the immune system in individuals leaving them vulnerable to disease and infection. These examples illustrate how animals undergoing important annual lifecycle periods are vulnerable and when exposed to a disruptive stimulus this can push the physiological capabilities of these animals to extremes that cause lowered fitness (Ellenberg et al. 2006; French et al. 2010).

The behavioural and physiological responses that a stressed individual displays do not always match and can lead to misleading interpretations of the individual's response to an introduced stimulus (Ellenberg et al. 2006; Weimerskirch et al. 2017). For example, Ellenberg et al. (2006), found that incubating Humboldt penguins showed no behavioural response to human presence but, using artificial eggs to record their heart rate, found that birds had significantly elevated heart rate. Similarly, Weimerskirch et al. (2017) found that incubating king penguins did not show any noticeable behavioural response to drone flights at altitudes close to 3 m but that their heart rate significantly increased when drones were at 10 m. These examples show individuals becoming cautious in response to a new stimulus but only invoke an 'anti-predator response' when near the potential threat (Ellenberg et al. 2006; Weimerskirch et al. 2017). This shows the importance to assess both behavioural and physiological responses of an animal to determine the initial point at which an individual becomes stressed by introduced stimuli.

1.3 Long-term impacts of human disturbance

The degree to which a species is impacted by an introduced stimuli can vary depending on the occurrence and intensity, the sensitivity of the individuals or the species, temporal and spatial factors, as well as other pressures they may be facing (Faeth et al. 2005; Chace & Walsh 2006; Lowry et al. 2011; Samia et al. 2015; Vincze et al. 2016). Short term impacts from introduced stimuli may alter short-term behavioural time budgets (Chan and Blumstein 2011) and elevate heart rates (Ellenberg et al. 2006), but may not necessarily have long-term fitness impacts on the individuals (Lowry et al. 2011; Samia et al. 2015; Vincze et al. 2016). If the individual however is undergoing a sensitive process, such as breeding or moulting, these short-term impacts can reduce its survival and reproductive success, which may have long lasting impacts on the entire population especially for small populations (Buxton et al. 2017a; Liu et al. 2020). For instance, Tern species are highly sensitive to disturbance (Melanie 2004) and a whole breeding colony of approximately 3000 elegant terns (*Thalasseus elegans*) has been shown to abandon its breeding site due to a drone crash landing in the middle of the colony, which resulted in > 1,500 eggs being lost (Thompson 2021). Such short-term disturbances can halt population growth and when combined with additional pressures, such as poor environmental conditions (Scopel & Diamond 2017) or reduced prey availability (Pink & Abrahams 2018), can induce population decline (Scopel & Diamond 2017; Pink & Abrahams 2018). Site abandonment of a species can reduce biodiversity within the area and disrupt the ecological processes that may depend on the daily activities of that species, which can lead to reduced energy flow throughout the trophic levels, interruption to nutrient cycles and indirect adverse impacts to surrounding species (Lee 1988; MacDougall et al. 2013).

Human disturbance can cause stressed individuals to develop chronic health complications (Barnett & Hemsworth 1990; Beans 1996; Calow & Forbes 1998; Patisaul & Adewale 2009). Prolonged physiological responses that occur outside the normal range that an individual can cope with can reduce their fitness (Calow & Forbes 1998). Stressed individual’s endocrine system can become activated invoking a “fight or flight response” with a short-term endocrine response, such as elevated heart rate and increase hormone production, all of which normally prepare the individual to take immediate action if needed (Barnett & Hemsworth 1990; Lowry et al. 2011; Derose-Wilson et al. 2015). However, long-term endocrine responses to reoccurring or permanent changes may cause substantial adjustments in both hormone and brain function systems, leading to adverse health impacts, such as ulcers, hypertension, arteriosclerosis, and suppression of the immune system (Barnett & Hemsworth 1990). The reoccurring presence of an endocrine disruptor can also reduce reproductive success (Patisaul & Adewale 2009) as demonstrated by Beans (1996) in bald eagles (*Haliaeetus leucocephalus*), where individuals had reduced reproductive output from consuming fishes contaminated with run-off pesticides. This led to feminized male embryos, weakened eggshells, and altered reproductive behaviours causing a significant decline in the species population (Beans 1996).

Light pollution can be a sign of permanent human presence within an area and its long-term establishment can influence permanent changes in surrounding species dynamics. For example, the physiology of greats tits (*Parus major*) was significant altered when exposed to artificial light, due to sleep deprivation which implicated their cognitive and metabolic processes, leading to reduced breeding and social behaviours, which in turn impacted species abundance and predator-prey interactions (Dominoni et al. 2022). Altered behaviours and fitness of a single species can cascade throughout the different trophic levels, leading to changes in resulting species biodiversity and abundance (Bennie et al. 2018; Owens et al. 2020). Therefore, the constant presence of artificial light in human developed areas can act as a selective tool altering the survival capabilities of the local wildlife (Ciach & Fröhlich 2017; Liu et al. 2020; Owens et al. 2020).

The introduction of anthropogenic noise can inhibit and disrupt individuals performing biologically important tasks and therefore render some habitats unsuitable (Reill et al. 2017). Similar to the long-term impacts of light pollution, anthropogenic noise can filter out certain species within an area creating a disturbed and unbalanced ecosystem which can reduce the efficiency of ecological processes and the overall health of the ecosystem. A study by Francis et al. (2011) monitored the distribution of avian species exposed to natural-gas extraction activities and found both a significant drop in avian species richness but also some disparities in how species responded to the activities: some species, such as the black-chinned hummingbird (*Archilochus alexandri*) and house finch (*Carpodacus mexicanus*), favoured noise polluted sites, while others, such as gray flycatchers (*Empidonax wrightii*), gray vireos (*Vireo vicinior*), black-throated gray warblers (*Dendroica nigrescens*), and spotted towhees (*Pipilo maculatus*), significantly nested farther away from noise polluted sites. Such alteration in species dynamics may force those that have fled into more degraded or overwhelming habitats or those that remain to experience increased pressures from sensory disturbance and increased predator presence (Lee 1988; MacDougall et al. 2013; Reill et al. 2017; Ciach & Fröhlich 2017; Liu et al. 2020; Owens et al. 2020).

1.4 Coastal developments and seabirds

Seabirds are important bio-indicators within marine environments and are commonly used by researchers to monitor pollutant levels and inform conservation efforts (Elliott et al. 1989; Vander Pol et al. 2009; Piña-Ortiz et al. 2016; Provencher et al. 2010; Rodríguez et al. 2017b; Baak et al. 2020). Seabirds are long-lived species that exploit a board range of habitats and are highly reliant on their senses for navigation, orientation, foraging, breeding, and avoiding predators (Boyle & Samson 1985; Parrish 2000; Tablado & Jenni 2015; Rodríguez et al. 2015; Blackwell et al. 2016; Blumstein 2016; Buxton et al., 2017a; Rajpar 2018). Therefore, the occurrence of rapid changes in seabird colony dynamics can indicate the presence of extremely harmful environmental conditions (Ryan 1988; Lloyd et al. 1974). Slow changes to their reproductive success and individual physiological health, on the other hand, can reveal that a persistent pollutant is present in the environment at low rates (Parrish 2000; Provencher et al. 2010; Baak et al. 2020). Seabirds differ from other non-pelagic bird species as they live most of their life at-sea and rely heavily on their senses, making them particularly vulnerable to disturbance from human activity (Friesen et al. 2017), with seabirds being the most threatened group of birds: 31 % of seabird species are globally threatened and 47 % of all seabird species have declining population trends (Dias et al. 2019).

Seabirds utilize two very important environments that are heavily used by humans. First seabirds forage in the marine environment, which largely overlaps with human resources extraction (Wiese et al. 2001; Higham & Shelton 2011; Quispe et al. 2020) and transportation activities (Hébert et al. 2006; Humphries & Huettmann 2014; Buxton et al. 2017b). Human activity in the marine environment can inhibit daily tasks undertaken by seabirds, as well as impact their health both directly and indirectly through collisions or being caught as by-catch (Merkel & Johansen 2011;

Rodríguez et al. 2017a) or from exposure to pollutants, such as plastics (Vikas & Dwarakish 2015; Van Franeker & Law 2015; Wilcox et al. 2015; Nations 2017) or chemical contamination (Henny et al. 1982; Diamond & Devlin 2003; Mallory & Braune 2012; Bianchini et al. 2022). Seabirds also use terrestrial habitats for roosting, breeding, and moulting (Parrish 2000; Rajpar 2018; Ganendran et al. 2015) where human disturbance can occur from recreational activities, urban or industrial developments (Beatley 1991; Small & Nicholls, 2003; Balk et al. 2009; Nations, 2017). Coastal developments can produce a lot of pollutants that negatively impact seabirds, such as plastic debris (Vikas & Dwarakish 2015; Van Franeker & Law 2015; Wilcox et al. 2015; Nations 2017), heavy metals or chemicals (Henny et al 1982; Vikas & Dwarakish 2015; Van Franeker & Law 2015; Wilcox et al. 2015; Nations 2017; Bureau, 2022), artificial lights (Ainley et al. 2001; Raine et al., 2007; Merkel & Johansen 2011; Rodríguez et al. 2012; Deppe et al. 2017; Rodríguez et al., 2017a), and anthropogenic noises (Ellenberg et al. 2006; Larcombe 2016; Dann & Chambers 2013, Buxton et al. 2017a).

Seabirds are highly prone to negative impacts caused by sensory disturbance, such as artificial light and anthropogenic noise, which can lead to reduced individual health, species abundance, and survival (Ainley et al. 2001; Ellenberg et al. 2006; Raine et al., 2007; Merkel & Johansen 2011; Rodríguez et al. 2012; Dann & Chambers 2013; Larcombe 2016; Deppe et al. 2017; Rodríguez et al., 2017a; Buxton et al. 2017a; Syposz et al. 2021). Artificial light emitted around coastal developments can attract nocturnal seabird species, such as petrels and shearwaters, causing them to become grounded in the illuminated areas (Rodríguez et al. 2017a). Grounded birds then become vulnerable to terrestrial threats, such as collisions with cars or predation (Rodríguez et al. 2015, 2017a; Syposz et al. 2021). Additionally, anthropogenic noise produced by tourists can result in seabirds increasing their vigilance behaviours (and thus spending less time in other critically important behaviours) and reduce their colony attendance (Buxton et al. 2017a, Franks 2017).

1.5 Thesis scope and objective

The majority of studies investigating the impacts of artificial light and anthropogenic noise on seabirds have focused on petrel and shearwater species (Ainley et al., 2001; Wiese et al. 2001; Soldatini et al. 2015; Derose-Wilson et al. 2015; Rodríguez et al. 2016, 2018, 2017a, 2017b; Buxton et al. 2017a; Franks 2017), with little focus on other species. Therefore, to better understand how different seabird species respond to sensory disturbance, it is important to investigate this question across other species. The aims of this thesis were therefore to investigate the short-term impacts of construction noises and artificial lights on little penguins (*Eudyptula minor*), to assess their population numbers and breeding success and to compare those to the back-faced cormorants (*Phalacrocorax fuscescens*) and the crested terns (*Thalasseus bergii*) as all three species are closely associated on Lipson Island (the study site selected for this study) and nest in the same habitat. These results will be used to better understand the implications that new coastal developments will have on these species.

1.6 Thesis structure

This thesis consists of five chapters. Chapter one provides some essential background information and knowledge needed to understand how sensory disturbance can negatively impact wildlife, and seabirds in particular.

Chapter two is a literature review that focuses on the pollutants produced by coastal developments and how common pollutants produced by these developments impact seabird species. The review differentiates between physical pollutants -- that produce unquestionably negative impacts on seabirds -- and sensory pollutants -- that induce mixed response across species and habitats. The chapter further highlights the need for additional assessment studies on the impacts of artificial light and anthropogenic noises on seabirds.

Chapter three investigates how little penguins responded to construction noises using playback experiments and whether this may affect their breeding success. In this chapter, little penguins were experimentally tested with broadcast of (1) construction and (2) rainfall noises during incubation. I predicted that individuals would display higher vigilance and distressed behaviours in response to the construction noises compared to the rainfall noises. I predicted that males and females will not differ in their behavioural and physiological responses based on previous studies in little penguins. I also predicted that breeding success will be lower for nests exposed to the noise experiments compared to those that were not.

Chapter four investigates the behavioural response of little penguins to artificial light. In this chapter, little penguins were experimentally exposed to artificial lighting when returning to their colony at night. I predicted that little penguins would significantly arrive earlier but that vigilance behaviours would decrease under artificially illuminated conditions. I also predicted that the occurrence of breeding behaviours would increase significantly under artificially illuminated conditions.

Chapter five summarises the findings from the two experimental chapters and discusses the implications of the results.

CHAPTER 2 – LITERATURE REVIEW

Seabirds are vulnerable to four common pollutants produced by coastal development and human activity, these include plastic debris (Provencher et al. 2010; Baak et al. 2020), chemical pollutants (Braune, 2012; Bianchini et al. 2022), artificial light (Rodríguez et al. 2017; Ginesete et al. 2017), and anthropogenic noise (Derose-Wilson et al. 2015; Franks 2017). Human presence in coastal areas is highly abundant with 14 out of the 17 largest cities in the world occurring in coastal areas where they provide humans with economical, logistical, and cultural benefits (Small & Nicholls, 2003; Balk et al. 2009; Nations, 2017). But the expansion and advancement of coastal developments exposes seabird colonies to the adverse impacts, such as the lowered fitness of individuals which can accumulate into the instability of seabird colonies, due to the release of these pollutants that can enter the environment through improper disposal and unfiltered release of sensory stimuli (Small & Nicholls 2003; Balk et al. 2009; Rodríguez et al. 2015; Nations 2017; Wilson et al. 2022). Plastic is a well-known pollutant in our oceans and in 2010 plastic waste produced from coastal developments equated to 99.5 million tonnes with 8 million tonnes ending up in the oceans due to mismanaged disposal (Ritchie & Roser 2018). Entanglement and ingestion are the main threats that can cause mortality or have sub-lethal effects leading to reduced foraging efficiency, breeding success, mobility, development, and increased vulnerability to predation (Thushari & Senevirathna 2020). Additionally, chemical pollutants enter the marine environment through high quantities of agricultural runoff and industrial effluents (Henny et al. 1982; Bureau 2022). Chemical contamination in seabirds has been reported to reduce reproductive, immunological, and neurological functions (Henny et al. 1982; Bureau 2022). Finally, sensory disturbance from artificial light and anthropogenic noise can alter behavioural time budgets and attention allocation in individuals resulting in lowered fitness due to individuals neglecting biological important tasks (Chan & Blumstein 2011; Siemers & Schaub 2011). This literature review will assess the impacts that each of these four pollutants have on seabirds, including any interactive and long-term impacts, and discuss how these adverse impacts have previously been managed.

2.1 Plastic pollution

2.1.1 Impacts to seabirds

Major water currents in our oceans facilitate the wide-spread distribution of plastic pollution, creating increased opportunity for seabirds to encounter such pollution (Ryan 1988; Provencher et al. 2010; Trevail et al. 2015; Tanaka et al. 2020). Plastic pollution is a worldwide issue with 44 % of seabird species estimated to be affected by marine debris ingestion and up to 1 million seabirds are killed each year as a result (Vikas & Dwarakish 2015; Van Franeker & Law 2015; Wilcox et al. 2015; Nations 2017). Plastics take over a century to degrade and the overproduction of plastic products and improper disposal has led to mass aggregations of plastic debris in our oceans (Vikas & Dwarakish 2015). While earlier studies found that seabirds may be more vulnerable to plastic pollution at the local levels (Ryan 1988; Trevail et al., 2015; Vikas and Dwarakish, 2015; Van Franeker and Law 2015; Wilcox et al. 2015; Nations, 2017), in recent years, plastic ingestions have been recorded in a wider diversity of seabird species and locations (Provencher et al. 2010; Wilcox et al. 2015; Baak et al. 2020). In support of this, Baak et al., (2020) found that 15 % of sampled black-legged kittiwakes (*Rissa tridactyla*) in the Arctic had plastic debris in their stomachs, which was the highest levels ever recorded for the species. Although there are still some safe havens free of plastic pollution, such as the Antarctic area where non-migratory species were found to have no plastic in their stomach during the breeding season (Van Franeker & Law 2015), the threat that plastic pollution poses on seabirds is set to intensify in future years considering the widespread distribution, pervasive, and rapid increase of plastic pollution in the marine environment (Wilcox et al. 2015).

One of the most harmful aspect of plastic pollution for seabirds is the ingestion of plastic debris (Ryan 1988; Roman et al. 2019), especially user plastics (Provencher et al. 2010; Baak et al. 2020). Yet ingestion rates can vary between species depending on their foraging behaviours and diets (Provencher et al. 2010; Baak et al. 2020). For example, Baak et al., (2020) examined plastic ingestion across four seabird species (northern fulmars *Fulmarus glacialis*, black-legged kittiwakes, thick-billed murres *Uria lomvia*, and black guillemots *Cepphus grille),* and found that murres and guillemots did not consume plastic debris as they are pursuit-diving seabirds and catch their prey below the surface, whereas 72 % of the northern fulmars and 15 % of the black-legged kittiwakes had plastic debris in their stomachs, likely due to the fact that they are surface-feeders that commonly mistake plastic debris for slow moving cephalopod and crustacean species (Baak et al. 2020). As a result, the most vulnerable seabird species to plastic ingestion are from the *Procellariiformes* order as majority of these species are surface feeders, primarily consuming squids (Provencher et al. 2010; Baak et al. 2020; Tanaka et al. 2020).

Seabirds are commonly documented to ingest plastic debris while foraging which leads to reduced individual fitness and breeding success due to inhibited nutrient intake (Ryan 1988; Provencher et al. 2010; Roman et al. 2019; Baak et al. 2020; Tanaka et al. 2020). Seabirds ingesting plastic debris consume less nutritional prey due to plastic loads filling up their stomachs and gizzards (Ryan 1988), which in turn lead to obstruction within their gastrointestinal tract, one of the most frequent cause of death from plastic consumption (Roman et al. 2019). Romain et al. 2019 showed that ingestion of one plastic item increase seabirds' chance of mortality by 20 %, that the ingestion of nine plastic items increases their chance of mortality by 50 %, and that ingestion of more than 90 plastic items results in 100 % mortality. The ingestion of plastic can also lead to reduced reproductive success (Ryan 1988). In his review across 60 seabird species, Ryan (1988) demonstrated that adults with higher plastic loads did not participate in the current breeding season or failed as breeders. If successful with their breeding, the plastic consumed by the breeding adults may also lead to intergenerational transfer, where breeding adults accumulate plastic particles and feed them to their chicks by regurgitation (Ryan 1988). Seabird chick survival rates are therefore reduced due to low nutrient intake and obstructions within their gastrointestinal tract (Romain et al. 2019).

Entanglement of seabirds from plastic debris is an additional adverse aspect of plastic pollution leading to reduced individual fitness as a result of restricted movements, inhibited prey consumption, or injury (Votier et al. 2011; Ryan 2018; Nisanth & Kumar 2019; Thushari & Senevirathna 2020). More than 50 % of seabird's species are vulnerable to plastic entanglement with the main culprit being fishing line and netting (Ryan 2018). The most vulnerable seabird species to entanglement are plunge-diving species (such as, terns *sulids*) that can become entangled with, for example bags around their necks, while foraging that can obstruct to their movement, vision or nutrient intake (Ryan 2018). Entangled individuals are also vulnerable to becoming stranded in an area due to the plastic also entangling onto a stationary object, resulting in starvation or attack from predators (Votier et al. 2011; Ryan 2018; Thushari & Senevirathna 2020). The degree in which plastic entanglement can restrict or harm an individual varies depending on the type of entanglement (Nisanth & Kumar 2019). For example, plastic debris wrapped around an individual's beak, neck or wings can restrict their movement and lower their foraging efficiency, while plastic debris wrapped around an individual's leg or single wing allows the bird to still fly and consume prey (Nisanth & Kumar 2019). The presence of plastic has become such a common occurrence within marine and coastal environments that some species of seabirds have begun using plastic debris as structural components within their nests. For example, Votier et al., (2011) found nesting northern gannets (*Morus bassanus*) showed a preference for using synthetic rope as nesting material which led to entanglement and deaths, particularly of chicks, and recorded nests containing on average ~470g of plastic with an estimated colony total of 18.5 tonnes (Votier et al. 2011). Therefore, plastic entanglement not only reduces adult fitness but has the potential to reduce breeding success in seabird species.

Plastic debris can be associated with chemical pollutants as toxic products stored in plastic remain on the surface of the debris and when ingested by seabirds can leach into their muscle tissue causing additional adverse impacts (Lithner et al. 2011; Thushari & Senevirathna 2020). This can lead to biomagnification of chemical pollutants, as well as plastic pollutants, by direct plastic ingestion or through ingested prey species that are already contaminated (Teuten et al. 2009). Seabirds can continue the flow of plastic pollution throughout the trophic levels when consumed larger predators (Romain et al. 2021). Also, the means in which plastic and chemical pollutants are present within the marine environment create a 'relayed effect' that increases the diversity of seabird species vulnerable to these pollutants. For example, plastics are reported to mainly impact surface-feeding seabirds that mistake plastic bags for their natural prey (Provencher et al. 2010; Tanaka et al. 2020; Baak et al. 2020) and chemicals largely impact deep diving seabird species that prey on fish species that can carry and transfer toxins from tissue and organs (Vander Pol et al. 2009).

2.1.2 Long-term impacts

The rate of plastic pollution is set to increase in future years thereby prolonging and intensifying the level of vulnerability for seabird species (Wilcox et al. 2015). Although, population decline in seabird species can be caused by multiple factors simultaneously (Provencher et al. 2010; Rodríguez et al. 2015; Thushari & Senevirathna 2020; Tanaka et al. 2020; Wilson et al. 2021), any increase in mortality rates can lead to cumulative effects that gradually lower seabird populations until they reach a threshold of which recovery is unlikely (Moloney et al. 1994; Roman et al. 2021). For example, mortality of juvenile wandering albatross (*Diomedea exulans*) due to plastic pollution, can lead to a time lag of 5 – 10 years before breeding populations become impacted and once population decline occurs in this species population growth rates can take 30 – 50 years to stabilize (Moloney et al. 1994). With several specific seabird species already highly impacted by plastic pollution researchers have documented that the diversity of species affected is increasing and reaching more remote colonies (Provencher et al. 2010; Baak et al. 2020). The addition of intergenerational transfer of plastic will also reduce fledgling survival rates (Ryan 1988) further halting the growth of populations to the point in which there will not be enough individuals to maintain a stable population (Moloney et al. 1994; Roman et al. 2021). Studies directly investigating long-term population trends of seabird species is limited, therefore there is a need for continued long-term monitoring of seabird colonies known to be highly exposed to plastic pollution to prevent population collapse.

2.1.3 Management plans

The proper management of plastic pollution is a global issue with the implementation of such plans being the responsibility of local governments to enforce (Lenzi et al. 2016; Thushari & Senevirathna 2020). Legalisation has been put in place to reduce the use of user plastics and implement more effective plastic waste management programs in many parts of the world (Ryan 1988; Cho 2009; Han et al. 2010; Thushari & Senevirathna 2020). At the global level, the UN Convention on the Law of sea provides an international legal framework for controlling plastic contamination, for governments to use as guidelines (Thushari & Senevirathna 2020). At the regional level, coastal clean-up programs have been completed around the globe, in areas of high plastic debris accumulation, along with discarded fishing gear collection projects that have been implemented in Hawaii and South African (Thushari & Senevirathna 2020). Local initiatives have been created to encourage individuals to properly dispose of plastic waste, such as fishing gear buyback programs that give incentive to properly dispose of fishing gear rather than dumping it in the ocean (Cho 2009; Han et al. 2010). Therefore, appropriate are being put in place around the globe to prevent the increase of plastic pollution leading into the oceans while also implementing clean-up programs to remove plastic debris from the environment.

2.2 Chemical pollution

2.2.1 Impacts to seabirds

Seabirds are top predators and as a result are more vulnerable to chemical contamination due to biomagnification (Diamond & Devlin 2003; Mallory & Braune 2012; Bianchini et al. 2022). Chemicals, such as heavy metals and chlorinated hydrocarbons, are persistent pollutants that become incorporated into the environment and are exchanged between trophic levels (Dureja & Rathore 2012). Chemical pollutants enter the marine environment through agricultural run-off, industrial effluent, and urban wastewater, where they are consumed by lower trophic level organisms and are absorbed into the lipids of the organism which facilitates the transfer of these toxins into higher trophic level species (Ohlendorf & Harrison 1986; Savinova et al. 1995; Dureja & Rathore 2012; Piña-Ortiz et al. 2016). Top predators, such as seabirds, are most vulnerable to consuming high levels of these chemicals as they consume a higher rate prey species which can then lead to adverse health complications and mortality (Diamond & Devlin 2003; Mallory & Braune 2012; Bianchini et al. 2022). Pesticides are one of the most harmful chemical pollutants and its unregulated used in the 1960’s led to a mass mortality event of guillemots (*Uria aalge*) along the west coast of Britain, deceased birds were found with high levels of organochlorine pesticide indicating the surrounding marine environment was highly polluted (Lloyd et al. 1974). This event had initiated the enforcement of regulations that restricted the use of organochlorine pesticides, leading to decreased levels within the environment, but researchers still use seabird species to monitor chemical contamination levels within the marine environment to prevent such a detrimental event from occurring again (Henny et al. 1982; Cifuentes et al. 2003; Voulgaris 2017; Bianchini et al. 2022).

Feeding habits, migratory behaviour and bird life strategy can influence the level of exposure seabirds have to pesticide accumulation (Tsygankov et al. 2018). For example, Tsygankov et al., (2018) found that northern fulmars had the highest concentration of pesticides detected in their internal organs (5874 ng/g lipid) due to their prey species (such as fish, fish eggs, shellfish, crustaceans and other invertebrates, carrions, and various fat wastes) having a high rate of organochlorine pesticide accumulation (Dureja & Rathore 2012; Shalini et al. 2020). Other seabird species in this study, such as crested auklet (*Aethia cristatella*), auklet-crumb (*Aethia pusilla*), and gray petrel (*Oceanodroma furcate*), had lower contamination levels due to their prey having weaker levels of pesticides (Tsygankov et al. 2018). Vander Pol et al., (2009) also found differing contamination concentrations with common and thick-billed murre eggs containing 2.5 times higher than glaucous gull (*Larus hyperboreus*) eggs. Different foraging strategies and habitats were thought to be a factor, where glaucous gulls are surface feeders that tend to utilize nearshore environments and murres are deep divers that feed primarily on fish in offshore environments up to 100-160 km from their nesting colonies (Vander Pol et al. 2009). Therefore, species that forage in the open ocean are more likely to encounter these chemical pollutants in higher trophic prey species that have accumulated more of these pollutants (Vander Pol et al. 2009).

Ingestion of toxic heavy metals can disrupt a seabird's endocrine system which can lead to reduced reproductive output (Payna et al. 2019). Several studies have found that the ingestion of chemicals can increase stress hormones in incubating parents leading to reduced parental care and lowered breeding success (Tartu et al. 2015; Blévin et al. 2018, 2020). Incubation-related behaviours and brood patch formation are influenced by hormonal regulation such as prolactin secretion and chemical contamination from the chlorinated pesticides can disrupt the production of these hormones resulting in lowered egg hatching probability (Tartu et al. 2015; Blévin et al. 2018, 2020; Sebastiano et al. 2021). Other studies have found that female seabirds can transfer toxins into their eggs and as a result lead to thinning of the eggshell, thinning by 18 – 20 % can lead to reduced hatching success (Henny et al. 1982; Bianchini et al. 2022). Gress et al., (1973) reported a colony of Californian cormorants (*Phalacrocorax penicillatus*) experienced near complete colony reproductive failure due to high contamination levels.

2.2.2 Long-term impacts

There is currently little evidence that chemical pollution causes significant population declines due to the complex interaction of pressures that seabirds are exposed it, which makes it difficult to detangle the true cause of population decline (Burger & Gochfeld 2002). Although, chemical contamination can cause eggshell thinning in some species, contamination rarely reaches critically levels that cause reduced reproductive success (Henny et al., 1982; Cifuentes et al., 2003; Bianchini et al., 2022). This could be due to seabirds having the ability to secrete toxins into their feathers, that they moult annually, and females secreting toxins into eggshells (Henny et al. 1982; Cifuentes et al. 2003; Bianchini et al. 2022). However, other studies suggested that chemical pollutants can alter the breeding behaviours of individuals through altered hormone production (Tartu et al. 2015; Blévin et al. 2018, 2020). But more detailed research is needed in this area to determine whether this is an intergenerational problem or a single generation problem.

2.2.3 Management plans

Stricter regulations have been put in place to prevent the use of harmful pesticides on agricultural properties in certain countries which has led to the successful reduction of toxic chemicals in the environment (Ryan 1988; Lloyd et al., 1974). For example. documented declines in chemical pollution followed severe restrictions on the applications of DDT, dieldrin, and heptachlor by legislative action in both Canada and the United States during the 1970s (Elliott et al. 1989; Cifuentes et al. 2003). Resulting in a general decline of organochlorine levels found in seabird eggs since the early 1970s (Elliott et al. 1989). Also in Norway, temporal trends have reported a decrease in legacy persistent organic pollutant concentrations in recent decades, with concentrations of detected pollutants appearing to not have a negative effect on seabird population development within the sampling area (Huber et al. 2015). Researchers use seabirds to monitor chemical pollutant levels in the surrounding marine environment with the analysis of eggshells being the most reliable method as contaminate levels would reflect recent exposure to chemicals (Henny et al. 1982; Cifuentes et al. 2003; Bianchini et al. 2022). Although, this is not an effective tool for all species as chemical exposure can be dependent on a species habitat, foraging behaviour, and diet (Vander Pol et al. 2009; Carravieri et al. 2021). But the continued monitoring of these pollutants in seabirds can help inform decision makers and create effective policies.

2.3 Light pollution

2.3.1 Impacts to seabirds

The impacts artificial light can have on seabirds can very between species and colony locations (Le Corre et al. 2002; Merkel & Johansen 2011; Rodriguez et al. 2014, 2017a, 2018; Syposz et al. 2021). A large diversity of petrel species has been documented to be attracted to artificially illuminated areas which causes individuals to become disorientated and colliding with structures or the ground, resulting in injury or morality (Ainley et al. 2001; Rodriguez et al. 2012, 2015, 2017a; Gineste et al. 2017). Studies have found that fledglings from these species that are undertaking their maiden flights are the most vulnerable to becoming grounded due to naïve attraction to the visual cues of artificial lights (Ainley et al. 2001; Rodriguez et al. 2012, 2015, 2017; Gineste et al. 2017). More fledglings are recorded grounded from colonies located near coastal cities where individuals must fly over the city in order to reach the sea, whereas individuals from colonies located on coastlines or offshore colonies are less likely to become grounded in artificially illuminated areas (Le Corre et al. 2002; Rodriguez 2014). Also, the longer a colony has been exposed to artificial light the increased rate of fledgling attraction to these sites over the years (Gineste et al. 2017). Although, Manx shearwaters (*Puffinus puffinus*) are found to be highly repelled by bright artificial white light with lowered individual activity under these conditions (Syposz et al. 2021). In contrasts, a study by Rodgrigues et al. (2018) found that little penguins preferred artificially illuminated paths due to the increased visibility, which may be helpful for this species as they forage during daylight hours due to their eye sight being highly dependent on light. Whereas nocturnal foraging species such as petrels, have a high sensitivity to light and thereby can pick up light sources across larger distances and when in proximity to light become blinded and disorientated (Le Corre et al. 2002; Rodriguez 2014, 2017a).

Majority of studies have focused on the frequency of bird groundings in artificially illuminated areas under certain environmental conditions, and mortality rates of seabirds but no studies have investigated the changes in behavioural times budgets in seabird species exposed to artificial light (Wiese et al. 2001; Merkel & Johansen 2011; Rodríguez et al. 2017a; Deppe et al. 2017). For example, Rodríguez et al. (2017a) noted that the number of grounded birds in illuminated areas was lowered on nights with a full moon, suggesting that the increase visibility allows seabirds more freedom to roam their environment without depending on localised sources of light. But in terms of colony attendance, breeding and social behaviour, this research is lacking and may provide more insight into the level of impact artificial light may have on seabird populations.

2.3.2 Long term impacts

The long-term impacts of seabird colony found near coastal developments with artificial light sources can vary (Ainley et al. 2001; Rodrigues et al. 2012; Gineste et al. 2017). For example, Gineste et al. (2017) found that a tropical shearwater (*Puffinus bailloni*) population remained stable over a 18 year period despite proximity to artificial light sources. Rodrigues et al. (2012) used number of fledgling groundings to monitor Cory's Shearwater (*Calonectris Diomedea*), Bulwer's Petrel (*Bulweria bulwerii)* and Macaronesian Shearwater (*Puffinus baroli)* over a 20-year period. In this time human population had increased, researchers found the number of rescued fledglings of Cory's Shearwater and Bulwer's Petrel increased and remained stable, respectively, whereas numbers of rescued Macaronesian Shearwaters sharply declined, indicating a worrying decline in the Macaronesian Shearwater's breeding population within the area (Rodrigues et al. 2012). Therefore, more long-term studies are needed on a wider range on seabird species in order to determine site-specific vulnerabilities of local breeding seabird colonies.

2.3.3 Management plans

The mitigation of artificial light is largely reliant on proponents to have efficient development designs that follow the avoidance, minimization, rehabilitation, and offset approach (Rodriguez et al. 2017b). This approach entails removing unnecessary forms of light, when this cannot be done, reducing intensity or duration of 'light on' periods. When individuals are impacted be artificial light provide rescue and rehabilitation programs to reduce mortality rates and where impacts from artificial light cannot be avoided nor minimized, using offsets to compensate which can include predator or invasive species control, restoration of degraded habitats, or supporting conservation actions within the local area (Rodriguez et al. 2017b). Independent groups have initiated rescue programs to retrieve grounded birds for rehabilitation and release (Telfer et al 1987; Gineste et al. 2017). Also, artificial light guidelines have been created to help inform proponents of safe designs that can mitigate adverse impacts to seabirds induced by artificial lights (Australia 2020), for example using shielded light to reduce fallout rates of seabirds around coastal developments (Telfer et al 1987).

2.4 Noise Pollution

Very little literature is available regarding the impacts that noise pollution has on seabird and most studies to date investigated the impact of human noise or presence (Dann and Chambers, 2013, Buxton et al., 2017, Ellenberg et al., 2006, Larcombe, 2016). Some studies however have monitored how seabirds respond to noises produced from aircraft, helicopters, drone, and boats (Brown 1990; Derose-Wilson et al. 2015; Soldatini et al. 2015; Bevan et al. 2018). For example, studies have found that aircraft flyovers have caused increased vigilance behaviours and flushing behaviours in breeding seabird colonies (Brown 1990; Derose-Wilson et al. 2015). Other noise experiments that have been conducted on seabird species range from drone flights (Bevan et al. 2018) to gunshot sounds (Labansen et al. 2021). All founding increased vigilance or elevated heart rate levels in seabirds exposed to a new noise but none have monitored the impacts of anthropogenic noise produced from construction and operation of coastal developments and whether the altered environment has caused a change in population dynamics.

2.4 Conclusion

Plastic pollution is the most well-studied and well-monitored of the four common pollutants produced from coastal developments with the least studied being noise pollution. We have also identified that plastic and chemical pollution cause unquestionably negative impacts to exposed individuals and therefore it is clear that reduction of these pollutants is necessary for the survival of seabirds, as well as other wildlife. Whereas light pollution cause invokes mixed responses between species, although most studies focused on petrel species, and mortality rates can vary depending on environmental conditions and individual age. While, for noise pollution it is unclear whether this poses a threat to seabird colonies and whether this introduced stimulus can cause alter behavioural time budgets or reduced fitness in seabirds. From this literature review we highlight the need for more research into the impacts of artificial light and anthropogenic noise, produced from coastal developments, on a wider variety of species.

CHAPTER 3 – NOISY NEIGHBOURS: EFFECTS OF CONSTRUCTION NOISES ON DECLINING LITTLE PENGUINS.

3.1 Introduction

Anthropogenic noise produced from coastal developments – such as buildings and infrastructure, transportation, and marine industries – as well as tourism and recreational activities is continuously expanding and altering the sensory environment thereby causing adverse impacts on wildlife (Jefferson et al. 2009; Lai et al. 2015). For example, sensory disturbance produced from coastal developments can invoke a stress response in animals, where their attention is consumed by the introduced stimulus and thus reduces the time, they spent performing biologically important behaviours, such as foraging, communicating, and breeding (Chan & Blumstein 2011; Derose-Wilson et al. 2015; Franks 2017; Bevan et al. 2018; Buxton et al. 2017a). Distracted animals may also be more susceptible to fatal attacks from predators due to their preoccupied state of mind not being able to detect and avoid threats (Chan et al. 2010). The continued expansion of human developments leads to more species being exposed to anthropogenic noise, and investigating individuals respond to such noise is essential in understanding the degree of impact.

Short-term exposure to introduced anthropogenic noise can lead to reduced long-term fitness impacts at the individual level (Barnett & Hemsworth 1990; Calow & Forbes 1998), which in turn can lead to population declines (Melanie 2004; MacDougall et al. 2013; Scopel & Diamond 2017; Pink & Abrahams 2018) or abandonment of disturbed sites (Bennie et al. 2018; Thompson 2021). In some cases, human presence deterred some mature seabirds from breeding in disturbed sites (e.g., Buxton et al. 2017a), while in other instances, a single short-term disturbance event caused an entire breeding colony to abandon their nesting site (e.g., Thompson 2021). When disturbance occurs during sensitive lifecycle processes, such as breeding or moulting, an individual's stress response is intensified and can lead to reduced reproductive success (Buxton et al. 2017a; Liu et al. 2020). For example, Brisson-Curadeau et al. (2017) found that experimental drone flights increased predation potential on thick-billed murre (*Uria lomvia*) eggs due to incubating adults initiating avoidance behaviours in response to drone proximity, leaving their nest unoccupied and giving opportunities for aerial predators to predate on their eggs. Reduction in breeding success can halt a species population growth and potentially lead to population collapse (Scopel & Diamond 2017; Pink & Abrahams 2018). Therefore, being able to identify when an individual has become disturbed due to an introduced stimulus, and before they experience long-term fitness costs, can better inform mitigation strategies.

When investigating how individuals respond to disturbance, there is increasing evidence that both behavioural and physiological responses need to be considered as some species or individuals may not display any anti-predator behaviours but still produce an internal stress response

(Weimerskirch et al 2002; Ellenberg et al. 2006; Soldatini et al. 2015). For example, Adelie penguins (*Pygoscelis adeliae*) increased their heart rate when approached within 15 m by a human without showing any modification of their behaviour (Giese 1998). Similarly, Weimerskirch et al. (2017) found no behavioural response in incubating king penguins (*Aptenodytes patagonicus*) to drone flights at altitudes close to 3 m but that penguins already showed increased heart rates when drones were at 10 m. High stress inducing events may cause substantial adjustments of both hormone and brain function systems, resulting in the formation of ulcers, hypertension, arteriosclerosis, and suppression of the immune system, all of which reduce an individual's fitness and ability to perform biologically important task to an optimal standard (Barnett & Hemsworth 1990). Therefore, if such undiagnosed periods of stress occur, they can lead to increase energy expenditure in individuals that may have long-term negative fitness consequences (Weimerskirch et al 2002; Ellenberg et al. 2006; Larcombe 2015).

Seabirds are the most vulnerable group of birds and play vital roles in ecosystem functioning processes within coastal habitats, such as trophic regulation, nutrient transportation, and community shaping (Graham et al. 2018). As a result, seabirds are often used as monitors of environmental repercussions caused from anthropogenic activities (Henny et al. 1982; Cifuentes et al. 2003; Votier et al. 2011; Voulgaris 2017; Ryan 2018; Thushari & Senevirathna 2020; Bianchini et al. 2022). Seabirds are highly dependent on their senses to navigate and orientate themselves, identify and capture prey, as well as identify suitable breeding sites or mates (Tablado & Jenni 2015; Blackwell et al. 2016; Blumstein 2016; Buxton et al. 2017a; Boyle & Samson 1985), making them highly sensitive to changes within their environment (Ryan 2008; Vander Pol et al. 2009; Piña-Ortiz et al. 2016; Rodríguez et al. 2017a; Mooney et al. 2019). The effects of noise pollution from coastal developments on seabirds has received little attention to date, with most studies focusing on the effects of human voices (Melanie 2004; Watson et al. 2014; Buxton et al. 2017a), boat or shipping traffic noises (Humphries & Huettmann 2014; Soldatini et al. 2015; Buxton et al. 2017b), and aircraft noises (Brown 1990; Hughes et al. 2008; Derose-Wilson et al. 2015; Bevan et al. 2018), and only one study investigating the relationship between ambient anthropogenic noise produced from coastal developments and vigilance response in Eurasian oystercatchers (*Haematopus ostralegus*) (Franks 2017).

In seabirds, response to human related-disturbance vary greatly between species (Buxton et al. 2017a; Culik et al. 1990) and sexes (Weimerskirch et al. 2002; Ellenberg et al. 2009). For example, Brandt's cormorants (*Phalacrocorax penicillatus*) increased their vigilance in response to human voices (Buxton et al., 2017a), while Adelie penguins do not (Culik et al. 1990). Similarly, great cormorants (*Phalacrocorax carbo*) use stationary wind turbine structures as roosting sites, while northern gannets (*Morus bassanus*) showed strong avoidance behaviour in sites with wind turbines (Dierschke et al. 2016). Although stress response in king penguins was not significantly different between male and females (Viblanc et al. 2015), a study by Wiemerskirch et al. (2002) also found that male wandering albatross (*Diomedea* exulans) exhibited a more intense response then feamles prior to handling. Similarly, Ellenberg et al. (2009) found that, when yellow-eyed penguins (*Megaduptes antipodes*) were exposed to human presence, females exhibited lower heart rates, but longer recovery times, compared to males, suggesting that males were more defensive but less stressed compared to females. Males in general tend to display more intense defensive behaviours, maybe due to higher hormone production (such as testosterone), (Kazama et al. 2011), while females tend to be more sensitive to disturbance at the nest due already existing stressors (linked to storing body reserves and building up the clutch) at this critical time (Goutte et al. 2010). Therefore, species- and sex-specific studies are important to better understand seabirds' response to human related-disturbances.

In this study, we investigated the behavioural and physiological response of little penguins (*Eudyptula minor*), an iconic Australian seabird, to introduced anthropogenic noise resulting from coastal development (specifically, construction noises). To do so, we tested the response of little penguins to experimental playback of construction and rainfall (control) noises on Lipson Island (South Australia). We chose construction noises as our disturbance stimulus because a port facility is expected to be built 1.5 km away from Lipson Island. As the study site is remote, surrounded by agricultural lands, and has not been previously exposed to major developments, it is important to determine how such disturbance will impact the locally breeding birds. We also investigated whether short-term exposure to construction noises would impact their breeding success to assess potential long-term impact of the proposed coastal development. To better understand potential factors affecting little penguin breeding success (see Pink & Abrahams 2018; Taylor et al. 2013; Scopel and Diamond, 2017), we also monitored the breeding success of black-faced cormorants (*Phalacrocorax fuscescens*) and crested terns (*Thalasseus bergii*), as all three species are closely associated on Lipson Island and nest in the same habitat. We initial aimed to include all three species in our playback experiment, but this was not possible due to change in timing (see results). Based on previous literature in birds and their response to anthropogenic noises (Brown 1990; Melanie 2004; Hughes et al. 2008; Watson et al. 2014; Humphries & Huettmann 2014; Soldatini et al. 2015; Derose-Wilson et al. 2015; Franks 2017; Buxton et al. 2017a, 2017b; Bevan et al. 2018), we predicted that individuals would display higher vigilance and distressed behaviours in response to the construction noises compared to the rainfall noises. We also predicted that females will not differ in their behavioural and physiological response based on previous studies in little penguins (Saraux et al. 2011, Carroll et al. 2016; Schaefer and Colombelli- Négrel 2021). We also predicted that breeding success will be lower for the nests exposed to the noise experiments compared to those nests that were not (Klomp et al. 1991, Weerheim et al. 2003, Giling et al. 2008).

3.2 Methods

3.2.1 Study Location

We conducted all our experiments and observations on Lipson Island Conservation Park (34°26'S, 136°26'E) located in Lipson Cove, 70 km north of Port Lincoln, South Australia (Figure 1). Lipson Island is a sandy island with a rocky coastline proclaimed as a Fauna Conservation Reserve in 1967 then renamed into a Conservation Park in 1972 (Birds SA, 2021). The island is 336 m long with a maximum width of 100 m and is located 250 m from the mainland beach of Lipson Cove (figure 3.1). The island is accessible to the public by foot at low tide during the summer months, yet no terrestrial predators (e.g., dogs, *Canis lupus familiaris*; cats, *Felis catus*; and foxes, *Vulpes vulpes*) can access the island (Robinson, 1996). The island is dominated by Nitre Bush (*Nitraria billardierei*) and Common Iceplant (*Mesembryanthemum crystallinum*) (Robinson, 1996), which covers majority of the island during the winter months and dies off during the summer months (Larissa Iasiello, pers. obs.). Numerous bird species have been recorded on Lipson Island, but the only species known to breed on the island include the black-faced cormorant, the pied cormorant (*Phalacrocorax varius*), the silver gull (*Chroicocephalus novaehollandiae*), the feral pigeon (*Columba livia domestica*), the crested tern, the sooty oystercatcher (*Haematopus fuliginosus*), the pacific gull (*Larus pacificus*), and the little penguin (Bird SA, 2021; Larissa Iasiello, pers. obs.). Other bird species of importance that have been recorded, but do not breed on the island, include the rock parrot (*Neophema petrophila*), the red-capped plover (*Charadrius ruficapillus*), the Australian pelican (*Pelecanus conspicillatus*), the white-bellied sea eagle (*Haliaeetus leucogaster*), and the Caspian tern (*Hydroprogne caspia)* (Bird SA, 2021; Larissa Iasiello, pers. obs.). During this study, we did our playback noise experiments on little penguins between May and August 2021 and surveyed little penguins, black-faced cormorants, and crested terns for population size and breeding success between April and November 2021.

3.2.2 Study species

(a) Little Penguins

Little penguins in Australia can breed anytime between April and March (Reilly and Cullen 1981; Johannesen et al. 2003; Colombelli-Négrel 2015). Breeding little penguins are central-place foragers, foraging 20-30 km to their breeding site during the day (Collins et al. 1999; Hoskins et al. 2008). Each breeding pair lays up to two eggs per clutch and up to two clutches per breeding season (Reilly and Cullen 1981; Kemp & Dann 2001; Johannesen et al. 2003; Colombelli-Négrel 2015). During incubation and the first two weeks of chick-rearing, the male and the female take turns to remain in the nest incubating the eggs or guarding the chicks while the other is out foraging (Miyazaki & Waas 2003). After the chick guarding period, both adults forage during the day, leaving the chicks unattended, and return after dark to feed their chicks until the chicks are 8-9 weeks old (Kemp & Dann 2001; Colombelli-Négrel 2015). The incubation period lasts 33-44 days, and chicks normally fledge ~8-9 weeks after hatching (Kemp & Dann 2001; Colombelli-

Négrel 2015). Little penguins are known to breed in a variety of nest types, such as surface nests (scrapes under an open bush), sand nests (nests dug in soft sand), bush nests (scrapes deep under a thick bush), rock nests (burrows under boulder or in rock crevices), or artificial nests (plastic boxes with rocks, metal drums or concrete structure) (see Marker 2016, Colombelli-Négrel 2019). The most common nest type on Lipson Island is sand nests (which represented 90 % of the nests found in this study), but some penguins also breed in rock nests (9% of nests found) and bush nests (1% of nests found). In May 2011, a two-day survey as part of Port Spencer's Public Environment Report recorded only 29 little penguins on Lipson Island (Madden-Hallet el al. 2011).

(b) Black-faced Cormorants

The breeding biology of the black-faced cormorant remains largely undocumented with little information regarding the timing of their breeding, the age at which chicks fully fledged and the frequency of breeding (Riordan & Johnston 2013). On Notch Island, located in the northern Bass Strait (Victoria), the peak egg laying period begins in July and continues until September, and is believed to be associated with air temperatures (Taylor et al. 2013). Other breeding colonies in Western Australia and South Australia have been documented as breeding between September and February (Taylor et al. 2013). In contrast, Riordan and Johnston (2013) monitored a breeding colony of black-faced cormorants at Outer Harbor, South Australia, and documented their breeding season between April and August. Cormorants lay their eggs asynchronously with copulation occurring throughout the breeding season (Nelson 2005). Clutch sizes can reach up to eight eggs, but the mean clutch size is often three eggs (Howell 2009; Nelson 2005). Black-faced cormorants construct their nests out of marine vegetation, such as seaweed, on top of rocky surfaces (Taylor et al. 2013). Incubation begins as soon as the first egg is laid, and both adults take turns incubating for approximately 30 days (Nelson 2005). Nestlings hatch asynchronously and brood reduction is known to occur due to competition between nestmates (Nelson 2005; Riordan 2015). In May 2011, surveys recorded 159 cormorants on Lipson Island (Madden-Hallet el al. 2011).

(c) Crested Terns

Crested terns gather at their breeding sites several months before breeding to engage in courtship behaviours (generally between September and December) (Langham & Hulsman 1986). Their breeding season has been recorded between September and January (Hooley 2016; Fromant 2020), with pre-laying behaviours occurring from September to November and egg laying beginning in November - December (Langham & Hulsman 1985). Crested terns breed in highly synchronised colonies (Mcleay 2010; Hooley 2016). Breeding pairs nest on any ground that is above high tide and devoid of tall vegetation; no nest is constructed as terns lay their eggs directly on the ground (Langham & Hulsman 1985). Females lay a single egg, which is incubated for approximately 28 days (McLeay et al. 2010). After hatching, the chicks are provisioned by both parents for approximately five weeks (McLeay et al. 2010). Crested terns are also considered central-place foragers and need healthy supply of prey species near their breeding site as they deliver one prey at a time to their chicks (McLeay et al. 2009b). In May 2011, surveys recorded two crested terns on Lipson Island (Madden-Hallet el al. 2011). The only other record of crested terns breeding on Lipson Island is from a photo of a large breeding colony of crested terns from Robinson et al. (1996).

3.2.3 Playback experiment – Little penguins only

We tested 33 adult little penguins from 26 different nests with the experimental broadcast of (1) construction and (2) rainfall noises. We conducted all playback experiments during the day (between 8am – 2pm local time) between May and August 2021; all experiments lasted two hours once started. None of the tested individuals had previously been captured for research purposes, but all individuals potentially had some past exposure to human presence due to the proximity of Lipson Island to the mainland and the fact that locals have been previously recorded on the island during summer (Corey Berryman, pers. comm.). Therefore, all tested individuals were expected to have a similar experience with humans, and differences in stress responses were not likely to be related to any prior manipulation. At the time of the experiment, only one adult was present in each of the tested nest.

For each tested nest, we first confirmed the (1) sex of the adult penguin present in the nest using photographic identification as previously described for little penguins (Colombelli-Negrel & Smale 2018; Schaefer & Colombelli-Négrel, 2021; Wasiak 2020) and that the bird was incubating eggs. We then added a dummy egg to the clutch without removing any of the natural eggs to record their physiological response (heart rate) following Schaefer and Colombelli-Négrel (2021). The dummy egg contained an internal omnidirectional lavalier condenser microphone (WL183, Shure Inc., USA) connected to a either a Zoom H4n or a Tascam DR-05 (TEAC Corporation, USA) as described in Schaefer and Colombelli-Négrel (2021). Once the dummy egg was placed, we set up a Sony AS20 Action Camera (Sony Corporation, Australia) directly at the entrance of the nest to record the adult behavioural responses. We also placed a small Moshi Bass burger speaker (Moshi Corporation, USA; sensitivity: > 80 dB: frequency response: 280 to 16 kHz) and ipod (Apple Inc., USA) approximately 50 cm away from the targeted nest and started a randomly chosen playback track. Once the experiment ended, we checked whether the incubating bird had accepted our dummy egg and collected all equipment.

Each track began with 45 minutes of silence to allow individuals to recover from the deployment of the equipment. Both Larcombe (2016) and Schaefer and Colombelli-Négrel (2021) showed that little penguin heart rate values returned to baseline levels within half an hour of deployment; hence we are confident that this interval was sufficient. The 45 minutes of silence included periodic short bursts (of 1 sec) of environmental noises (noises caused by the wind and the ocean and obtained from previous recordings, played at ~ 65 db at 1 m every 5 minutes) to prevent the iPod from shutting down. This was followed by (1) five minutes of either continuous construction or rainfall noises, (2) another 45 minutes of silence (to allow recovery from the first playback), (3) another five minutes of playback (if construction noises were played first then the rainfall noises were played second and vice versa), and (4) another 30 minutes of silence. We created ten different tracks (five starting with the construction noises and five starting with the rainfall noises). To avoid pseudo-replication, we used different noises for each of the playback tracks. We obtained rainfall and construction noises as waves files from different websites (rainfall noises: https://mixkit.co/free-sound-effects/rain/ and https://www.shockwave-sound.com/; construction noises: http://free-loops.com/4598-construction-sounds.html, https://www.wavsource.com/sfx/sfx.htm, and https://www.mediacollege.com/downloads/sound-effects/machinery/). All the construction noises included general construction noises, such as vehicles, banging and a jackhammer. The rainfall noises were played at ~ 60-65 db (measured at 1m) and the construction sounds were played at ~70 db (measured at 1m) with peaks of ~75-80 db when a jackhammer was used.

From the video recordings, we first confirmed our field observation that the individual had accepted the dummy egg (or not) and then measured: (1) ‘behavioural recovery from deployment’ (total time in seconds taken by the individuals to stop showing signs of vigilance or distress behaviour after the deployment of the camera and dummy egg); (2) ‘baseline vigilance’ (total time in seconds spent in vigilance during the five minutes immediately before the beginning of the first or second playback); (3) ‘baseline distress’ (total time in seconds spent in distress behaviour during the five minutes immediately before the beginning of the first or second playback); (4) ‘latency’ (total time in seconds taken by the individuals to respond to the stimuli, which was measured as the time between the beginning of the playback and the first evidence of vigilance or distress behaviour); (5) ‘playback vigilance’ (total time in seconds spent in vigilance during the five minutes of playback); (6) ‘playback distress’ (total time in seconds spent in distress behaviour during the five minutes of playback); and (7) ‘behavioural recovery from playback’ (total time in seconds taken by individuals to fully stop showings any signs of vigilance or distress behaviours once the playback started). To avoid biases, the same observer recorded all penguin behaviours. An individual was recorded as vigilant if it was scanning the environment, generally with its head upright moving quickly from right to left and its neck extended and displaying intense staring (Fowler 1999; Holmes et al. 2005; Sherwen et al. 2015; Schaefer and Colombelli-Négrel 2021). An individual was recorded in distress if it was shivering continuously (either just is wings or its entire body), breathing heavily, showing pupil dilation (when visible), and/or appeared completely frozen in an unnatural position (Holmes et al. 2005; Ellenberg et al. 2012; Schaefer & Colombelli-Négrel 2021). We calculated ‘vigilance intensity’ as baseline vigilance - playback vigilance.

From the heart rate recordings, we measured: (1) ‘heart rate recovery from deployment’ (total time in seconds taken by the individual to reach baseline heart rate ± two standard deviations following the deployment of the camera and dummy egg); (2) ‘baseline heart rate’ (average heart rate in beats per minute during the five minutes immediately before the beginning of the first or second playback); (3) ‘heart rate playback’ (average heart rate in beats per minute during the five minutes of playback); and (4) ‘heart rate recovery from playback’ (total time in seconds taken by the individual to return to baseline heart rate ± two standard deviations once the playback started). To avoid biases, the same observer manually scored all heart rates. Heart rate was manually counted for each full minute or for 30 seconds and extrapolated into a minute when recordings were interrupted following Ellenberg et al. (2006, 2012), Larcombe (2016) and Schaefer and Colombelli-Négrel (2021). We averaged heart rate for five minutes during baseline and playback and calculated ‘heart rate intensity’ (baseline heart rate - playback heart rate)

To determine the potential impact of the playback experiments on hatching and breeding successes, we monitored 67 nests (26 used in the noises experiments and an additional 41 not used in the noises experiments) every three weeks and noted the number of eggs, chicks and adults present in each nest during each visit. Following Colombelli-Négrel (2015), we defined hatching success as the number of eggs that hatched compared to the number of eggs found and considered eggs abandoned if they felt cold to touch and were unattended for two consecutive visits. We defined breeding success as the number of chicks that fledged per breeding pair and were not found depredated nor in any other nests (Colombelli-Négrel 2015). Predation was scored as suspected if chicks were found dead outside the entrance of the nest with clear signs, such as torn off appendages (Colombelli-Négrel 2015; Larissa Iasiello pers. obs.). Chicks that disappeared before the age of 7 weeks or died without those signs were considered to have died due to poor growth or reduced food supply from parents (Colombelli-Négrel 2015).

3.2.4 Breeding success

(a) Little penguins

To determine the number of active penguin nests and breeding success in little penguins, we visited Lipson Island approximately every three weeks between April and November 2021. During each visit, we actively searched the island in its entirety for active little penguin nests. Penguin nests were identified as active if they contained egg(s), chick(s) or adult penguin(s) (Schumann et al. 2013; Colombelli-Négrel 2015, 2017). We marked each active penguin nest found with a small flagging tape attached to existing vegetation next to the nest and gave it a unique nest ID. We also recorded the GPS coordinates for each nest using a Garmin GPS 64 (Garmin Ltd, Australia); however, due to the poor accuracy of the GPS data points obtained, it was more reliable to mark out the nests manually. We recorded a nest as successful if at least one egg hatched and one chick fledged (i.e., it disappeared from the nest at approximately eight weeks of age and was not found depredated nor in any of the other nests) (Colombelli-Négrel 2015). We recorded the number of breeding attempts per breeding pairs and defined breeding success as the number of chicks that fledged per breeding pair (Colombelli-Négrel 2015).

(b) Black-faced cormorants and crested terns

To determine the breeding success of black-faced cormorants, we visited Lipson Island eight times between the beginning of April 2021 and the end of August 2021. During each visit, we used a Samsung S21+ (Samsung Inc, Australia) mobile phone camera to take photos and videos of the entire breeding colony from various viewpoints 5 m away from the colonies so that we did not disturb the birds or cause them to abandon their nests. From these photos and videos, we then manually counted the number of nests, nestlings, juveniles and adults following Terletzky and Ramesy (2014). An active nest was defined as a concave shape with built up edges capable of holding eggs and occupied by at least one adult bird and/or nestling(s) (Taylor et al. 2013). A recently abandoned nest was defined as a concave shape with built up edges capable of holding eggs, but with no adult, egg, or nestling present (Taylor et al. 2013). Cormorant nests are destroyed by strong winds when left unattended and therefore we could confidently confirm that empty nests were made and abandoned in the current breeding season. Nestlings were identified as chicks inside a nest with either naked, black coloured skin or grey skin with black-brown plumage and dark brown eyes (Riordan & Johnston 2013). Juveniles were identified as chicks that had successful fledged from their nest and had brown-white speckled plumage and pale eyes (Riordan & Johnston 2013). Adults were identified as individuals with black upper parts and white breast plumage with black eye feathering and blue-green eyes (Riordan & Johnston 2013). The same observer reviewed all the photos and videos to count the number of nests, nestlings, juveniles, and adults.

We used the same survey method as described above to survey the crested tern colony on Lipson Island and continuously monitored for evidence of roosting crested terns every three weeks between the beginning of April until the end of November 2021. We identified courtship behaviours as single individuals walking with their wings flexed out, their necks erected, and their heads constantly flicking up and down or as couples or trios taking off in courtship flights (Dunlop 1987; Larissa Iasiello, pers. obs.). We confirmed egg laying by observing individuals sitting directly on the ground and incubating an egg (Dunlop 1987).

3.2.5 Statistical Analysis

We used SPSS version 25.0 for Windows (SPSS Inc., Chicago, IL, U.S.A) for all statistical analyses. Data are shown as mean ± standard deviation unless otherwise stated. Prior to analysis, we assessed collinearity between (1) continuous predictors using the variance inflation factors (VIF) analysis or (2) categorical predictors using Chi Square and Pearson’s correlations. In all final models, VIF values were well < 2, Chi square values were > 0.05, and Pearson’s coefficients were close to zero confirming no collinearity (Zuur et al. 2009; Fox et al. 2015).

We examined ‘behavioural recovery from deployment’ using a General Linear Model (GLM) with a normal distribution and an identity link function with ‘sex’ (*male, female*) and ‘egg acceptance’ (*whether the individual accepted the dummy egg or not*) as fixed factors. The model did not converge when ‘individual ID’ was used as a random factor and therefore we removed it from the analysis.

We converted ‘latency’ (*time taken for individual to show first signs of vigilant/distress behaviour from start of playback*) into a binary variable categorised as: fast (responded to playback within < 5 seconds) or slow (responded to playback within > 5 seconds) and analysed it with a Generalised Linear Mixed Model (GLMM) with a binomial distribution and a logit link function. We analysed ‘vigilance Intensity’ (*the difference in the time spent in vigilance between the playback period and the baseline* period) and ‘behavioural recovery from playback’ using GLMMs with a normal distribution and an identity link function. As very few individuals showed signs of distress during the playback, we created a categorical distress response based on whether the individuals’ showed signs of distress during the playback (*yes, no*) and used GLMM with a binomial distribution with a logit link function. For each of these models, we used ‘sex’, ‘playback type’ (*construction, rainfall*), ‘playback order’ (*first, second*) as fixed factors and ‘individual ID’ as a random factor. The models did not converge when ‘nest ID’ was used as a random factor and thus we removed it from the analyses.

We analysed ‘heart rate recovery from deployment’ using GLMM with a normal distribution and an identity link function with ‘sex’ and ‘time taken to accept dummy egg’ (*time taken by the individual to accept the dummy egg and sit directly on top of the egg*) as fixed factors and ‘individual ID’ as a random factor. The model did not converge with ‘individual ID’ as a random factor and therefore was removed from the analysis. We examined ‘heart rate intensity’ using GLM with a normal distribution and an identity link function with ‘sex’, ‘playback type’, and ‘playback order’ as fixed factors. The model did not coverage when ‘nest ID’ and ‘individual ID’ were used as random factors and therefore we removed them from these analyses. We categorised ‘heart rate recovery from playback’ into fast (*within five minutes*) or slow (*over five minutes*) and used GLMM with a binomial distribution and logit link function with ‘sex’, ‘playback type’ and ‘playback order’ as fixed factors and ‘individual ID’ as a random factor. The model did not converge when ‘nest ID’ was used as a random factor and thus it was removed from the analysis.

We categorized a nest as having a successful breeding attempt if they produced at least one fledgling for the analyses. We examined whether exposure to the playback experiment predicted ‘breeding success’ using a GLM with a binomial distribution and a logit link function with ‘nest tested’ (*whether nest was exposed to playback experiment*) and ‘number of tested nests within 15 m radius’ as fixed factors. The model did not converge with ‘nest ID’ as a random factor and therefore it was removed from the analysis.

3.3 Results

3.3.1 Playback experiment – Little penguin only

3.3.1.1 Recovery from deployment

Not all the tested 33 nests could be used in the analysis due to equipment errors, poor visibility, bird not accepting the dummy egg or not sitting on the egg for the total length of the experiment. Therefore, the final sample sizes for the behavioural and physiological response are n = 22 and n = 15, respectively.

Mean 'behavioural recovery from deployment' was 12 ± 9 minutes (range of 2 – 35 minutes). None of the fixed factors significantly correlated with 'behavioural recovery from deployment' (GLMM - 'sex': $F_{1,19} = 0.35$, $p = 0.56$; 'egg acceptance': $F_{1,19} = 1.32$, $p = 0.27$; Table 1). The random factor 'nest ID' however significantly correlated with 'behavioural recovery from deployment' (Wald's Z = 2.72, $p = 0.007$)

Mean 'heart rate recovery from deployment' was 27 ± 7 minutes (range of 18 – 38 minutes). None of the fixed factors significantly correlated with 'heart rate recovery from deployment' (GLMM - 'sex': $F_{1,9} = 0.28$, $p = 0.61$; 'time sat on egg': $F_{1,9} = 0.49$, $p = 0.50$; Table 1).

3.3.1.2 Response to playback

Mean 'latency' to respond to the playback was 7 ± 14 seconds (construction noises: 3 ± 4 seconds, n = 22; rainfall noises: 11 ± 21 seconds, n = 17). None of the fixed or random factors correlated with 'latency' (GLMM - 'sex': $F_{1,35} = 1.73$, $p = 0.20$; 'playback type': $F_{1,35} = 1.87$, $p = 0.18$; 'playback order': $F_{1,35} = 1.43$, $p = 0.240$; 'individual ID' : Wald's Z = 0.68, $p = 0.50$; Table 1).

Individuals spent significantly more time in vigilance (55 seconds more) during the construction noises playback compared to the rainfall playback (GLMM: $F_{1,35} = 14.31$, $p = 0.001$; Table 1, Figure 2). None of the other factors correlated with 'vigilance intensity' ('sex': $F_{1,35} = 2.89$, $p = 0.10$; 'playback order': $F_{1,35} = 0.05$, $p = 0.82$; 'individual ID': Wald's Z = 1.68, $p = 0.09$; Table 1).

None of the factors correlated with the occurrence of distress behaviour (GLMM - 'sex': $F_{1,35} = 2.47$, $p = 0.12$; 'playback type': $F_{1,35} = 2.69$, $p = 0.11$; 'playback order': $F_{1,35} = 3.32$, $p = 0.08$; 'individual ID': Wald's Z = 0.81, $p = 0.42$; Table 1).

Mean 'heart rate response' to the playback was 2 ± 11 bpm (construction noises: 5 ± 14 bpm, n = 13; rainfall: 2 ± 7 seconds, n = 12). None of the fixed factors correlated with 'heart rate response' to playbacks (GLM - 'sex': $F_{1,21} = 0.19$, $p = 0.67$; 'playback type': $F_{1,21} = 2.78$, $p = 0.11$; 'playback order': $F_{1,21} = 0.51$, $p = 0.48$; Table 1).

3.3.1.3 Recovery from playback

Mean 'behavioural recovery from playback' was 7 ± 9 minutes (range: 3 seconds – 36 minutes). Individuals took significantly longer time to recovery (4 minutes longer) from the construction playback than from the rainfall playback (GLMM: $F_{1,35} = 6.06$, $p = 0.02$; Table 1, Figure 3). 'Playback order' also significantly correlated with 'behavioural recovery from playback' with individuals taking significantly longer to recover (4 minutes longer) when exposed to the construction playback first ($F_{1,35}$ = 6.68, p = 0.01; Table 1). The remaining factors did not correlate with 'behavioural recovery from playback' ('sex': $F_{1,35}$ = 0.59, p = 0.45; 'individual ID': Wald's Z = 1.64, p = 0.10; Table 1).

Mean 'heart rate recovery from playback' was 7 ± 9 minutes (range: 5 – 28 minutes). None of the factors correlated with 'heart rate recovery from playback' (GLMM - sex': $F_{1,21}$ = 1.30, p = 0.27; 'playback type': $F_{1,21}$ = 0.91, p = 0.35; 'playback order': $F_{1,21}$ = 0.15, p = 0.71; 'individual ID': Wald's Z = 0.12, p = 0.91; Table 1).

3.3.1.4 Hatching and breeding success

Breeding success for little penguins on Lipson Island was 1 ± 1 fledglings per breeding pair (n = 66 breeding pairs). Exposure to the noise experiment significantly predicted 'breeding success', with nests exposed to the noise experiment being more likely to be successful (to produce at least one fledgling) compared to those not exposed to the experiment (GLM: $F_{1,26}$ = 7.53, p = 0.01; Table 1, Figure 4). The 'number of nests tested within 15 m' did not predict 'breeding success' ($F_{1,26}$ = 0.18, p = 0.67; Table 1).

3.3.2 Breeding Success of Little Penguins, Black-faced Cormorants and Crested Terns

(a) Little penguins

We found a total of 66 active little penguin nests between April 2021 and November 2021, which estimates the population size to 132 breeding individuals. Nests were aggregated within the vegetated centre of the island (see Figure 1), where we recorded 58 sand burrows and eight rock nests. In total, 148 eggs were laid, from which 88 chicks hatched and 73 fledglings fledged. Out of the 66 nests, we recorded at least one failed breeding attempt for 34 nests/breeding pairs: for 9 nests (26%), the eggs were abandoned; for 10 nests (29%), only one chick hatched, and for 3 nests (9%), chick(s) were found dead outside the nest. Out of the 66 nests, 32% showed evidence of a second clutch. Figures 5a and 5b provide some evidence of little penguins breeding on Lipson Island.

(b) Black-faced cormorants

Black-faced cormorants already had commenced breeding during our first trip on the 17th April 2021, with 70 breeding adults present and sitting on their nests and 91 empty nests. We recorded no juveniles during this trip. Cormorant's nests were located on the northern tip of the island (Figure 1), and nests were all made from vegetation, such as seaweed. Out of the 70 remaining nests, 32% (n = 22) had incubating adults sitting on either eggs or newly hatched nestlings, while the remaining 68% (n = 48) had at least one adult standing next to 1 or 2 nestlings with brown feathers. On the 15th May 2021, all nests were empty and we recorded 47 – 50 juveniles aggregating along the rocky coastline line. The juveniles remained on the island until August 2021 when they had grown all their adult feathers. Figures 5c and 5d provide some evidence of black-faced cormorants breeding on Lipson Island

(c) Crested tern

Since the first trip in April 2021, we spotted a group of crested terns (approx. 100 – 200 individuals) roosting on the southern point of Lipson Island and on the adjacent mainland rocky coastline. Courtship behaviours began in early October with a noticeable increase in crested tern numbers (approx. 263 individuals). By late October, crested terns had aggregated on the northern centre of the island to breed (Figure 1*)*. We counted approximately 195 breeding crested terns but the number of eggs at the time was unknown. On the 31st of October, all crested terns abandoned their nests and eggs for unknown reasons. A week later, we found remnants of approximately 36 crested tern eggs that had been predated upon by silver gulls (*Chroicocephalus novaehollandiae*) and only 10 – 20 adult crested terns were seen on the mainland beach of Lipson Cove. The tern population returned to 100 – 200 individuals by January 2022, but no second breeding attempt was observed (Corey Berryman, pers. comm.). Figures 5e and 5f provide some evidence of crested terns breeding on Lipson Island.

3.4 Discussion

The effects of noise pollution from coastal developments on seabirds has received little attention to date (Melanie 2004; Watson et al. 2014; Franks 2017; Buxton et al. 2017a), therefore there is a need to further understand the behavioural and physiological response of exposed seabird species to coastal developments. In this study, we found that little penguins significantly increased their vigilance and took longer to stop exhibiting such vigilance when exposed to construction noises compared to rainfall noises (our control). However, they showed no difference in heart response to the playback. We also found that playback experiments significantly impacted breeding success, with nests exposed to the playback being more successful than those that were not. Our results suggest that exposure to an anthropogenic noise can cause individuals to become more alert but do not perceive the stimulus as a substantial threat.

We found that little penguins increased their vigilance and took longer to stop exhibiting such vigilance when exposed to construction noises. Therefore, the behavioural response of little penguins to construction noise did initiate a short-term response but was not perceived as a substantial threat (Larcombe 2016). The stimulus still caused an individual to become distracted with a clear deviation in their attention and brain function towards displaying anti-predator behaviours (Laughlin 2001; Chan & Blumstein 2011). Though the disruption during our short-term experiment is not believed to have significantly impacted the daily behavioural time budget of individuals but repeated reoccurrence of an introduced stimulus, such as construction noise, is likely to reduce the time an individual spends preforming self-care behaviours (Fanning et al. 2020). Also, during our breeding surveys we noticed that adults were a lot more defensive when their chicks had hatched, therefore future studies may seek to investigate the behavioural response of individuals throughout the breeding period to determine at what stages of breeding lead to the most intense response.

Our study did not find a significant elevation in heart rate of little penguins exposed to construction noise although they did have a significant increase in vigilance behaviours. Similarly, Derose-Wilson et al., (2015) found Wilson's plovers (*Charadrius wilsonia*) increased vigilance behaviours when aircrafts flew overs but it did not cause a significant increase in heart rate. Whereas, wandering albatrosses almost doubled their heart rate in response to human presence and took 2-3 hours to recover from the disturbance (Weimerskirch et al. 2002). A study by Larcombe (2016), found that little penguins displayed a more intense response when there was direct disturbance by researchers to the individuals and less of a response to playback recordings. Therefore, the lack of an accompanying visual threat in association with the introduced noise may have caused a less intense response of little penguins to the construction noise in our study (Ellenberg et al. 2006; Derose-Wilson et al. 2015; Larcombe 2016). For example, Colombelli-Négrel & Katsis (2021) subjected little penguins to standardized nest intrusions which invoked aggressive, anti-predator behaviours from disturbed individuals.

While studies on other seabird species have documented sex-differences in seabirds in response to disturbance (Weimerskirch et al. 2002; Ellenberg et al. 2009), we found no correlation between the behavioural or physiological responses of individuals of different genders, which supports previous work on little penguins (Saraux et al. 2011; Carroll et al. 2016; Scheafer & Colombelli-Négrel 2021; Colombelli-Négrel & Katsis 2021). It has been suggested that individual personality (Saraux et al. 2011; Larcombe 2016; Ellenberg et al. 2009; Colombelli-Négrel & Katsis 2021) as well as previous experience with human disturbance (Ellenberg et al. 2009; Larcombe 2016; Colombelli-Négrel & Katsis 2021; Schaefer & Colombelli-Négrel 2021) may play a more important role than sex in little penguins when responding to disturbances. For example, Schaefer & Colombelli-Négrel (2021) found that little penguins breeding in colonies located in highly disturbed areas had more intense behavioural and physiological responses to playback experiments than those breeding in less disturbed colonies. Therefore, future studies should incorporate within their experimental design the recording of more unique traits that may influence an individual's response to disturbance.

The disturbance of individuals at this small-scale is not likely to consume large amounts of their daily energy budgets, with a study by Larcombe (2016) revealing that an extreme physiological stress responses in little penguins, initiated by researchers weighing individuals, took > 31 minutes to recover and used 2.8 % of their daily energy budget. Though this once off occurrence may not have detrimental impacts to individual fitness, repeated exposure and additional pressures that may reduce or inhibit energy intake activities, can then lead to reduced fitness (Faeth et al. 2005;

Chace & Walsh 2006; Lowry et al. 2011; Samia et al. 2015; Vincze et al. 2016; Buxton et al., 2017a; Liu et al., 2020). but the distraction caused by an introduced noise can lead to the individual being vulnerable to other threats, such as predation (Ellison & Cleary 1978; Buxton et al. 2017a), and reduces the time spent exhibiting daily tasks such as resting or preening (Chan & Blumstein 2011; Fanning et al. 2020).

Our study found that exposure to the nests exposed to the noise experiment had a higher breeding success rate compared to the nest that were not exposed. This could be a bias result as the nests that we chose for the noise experiment were easily accessible and had a clear line of vision for the ease of data collection. While majority of the nests that were not used for the noise experiment had obscured lines of vision where the burrows were too deep and the penguins were located around a curve. This meant we could not reliably monitor breeding success for these burrows. But due to the noise experiment not resulting in a significant decrease in breeding success for individuals exposed to construction noise this could indicate no disruption to incubation behaviours of little penguins. Though additional studies should investigate parental care of individuals exposed to the noise stimulus during later breeding periods (Frid & Dill 2002). In addition, the little penguin colony on Lipson Island colony was significantly higher (n = 132) than previous surveys (n = 29, Madden-Hallet et al. 2011). While breeding success was high (1.11 ± 1.17 fledglings per breeding pair) and consistent with other South Australian colonies (ranging from 0.6 to 1.2 fledglings per breeding pair; Colombelli-Négrel 2018; Johnson & Colombelli-Négrel 2021), due to a lack of previous monitoring on the island, we are unable to determine whether such breeding success is indicative of the productivity of the colony. Therefore, continued monitoring of this colony is needed to establish accurate population records, especially in light of a new port facility being constructed just 1.5 km away from the colony.

Our surveys of the black-faced cormorant colony on Lipson Island found that they began breeding earlier than expected with previous studies stating that black-faced cormorants largely breed during the winter months, during periods of minimal air temperatures (Riordan & Johnston 2013; Taylor et al. 2013). Another interesting founding was that majority of nests were only seen with one chick (maximum of 2), surprisingly small compared to previous studies that document black-faced cormorants having a mean clutch size of 3 (Riordan & Johnston 2013). Cormorants have been documented to be opportunistic species, with their breeding behaviours reflecting environmental conditions (Ellison & Cleary 1978; Taylor et al. 2013). Silver gulls also had an overlapping breeding season with the cormorants on Lipson Island, and as gull species are known to predator on eggs and chicks from other cormorant species (Ellison & Cleary 1978) and was also witnessed during field surveys in this study. Gull predation may have occurred and led to lowered clutch sizes and increased abandoned of nests (Ellison & Cleary 1978). Ellison and Cleary (1978), noted that naïve colonies of double crested cormorants to human disturbance showed a stronger tendency to abandon their nests, allowing for gull predation, compared to the following breeding season where adults showed more aggressive, protective behaviours when humans approached their nests (Ellison & Cleary 1978). Therefore, the lower and infrequent human disturbance on Lipson Island on black-faced cormorant colonies may make them more skittish when the public does venture to the island leading to a higher predation rate and lowered reproductive output of the colony. Overall, there is limited research available on the breeding biology of black-faced cormorants and therefore it is unknown if the low clutch sizes and on-set of breeding was significantly different compared to other breeding colonies or previous breeding seasons on Lipson Island. Continued monitoring of the Lipson Island colony is needed to ensure the cormorants are not in a declining trend which would call for conservation measure can be put in place, as the development of a new port facility would only intensify the declining trend.

The crested tern colony on Lipson Island is believed to have significantly decreased with a previous survey estimating the breeding colony to be approximately 2000 individuals in 2006 (McLeay et al. 2009a). Locals have also noticed the decline in crested tern numbers over the years suggesting increased human and dog presence on the mainland beach of Lipson Cove being responsible for this (Corey Berryman, pers. Comm.). Tern species are highly sensitive to human disturbance, especially during earlier phases of breeding (Melaine 2004; Bevan et al. 2018; Thompson 2021). In our study, we documented the on-set of breeding in October 2021. While surveying the breeding colony we did unintentional initiate two flushing events but the colony settle within a minute, but footage from 7pm that night reveals the crested tern colony completely abandoning their nesting site. The cause of nest abandon is unknown but previous studies have documented colony disturbance (Bevan et al. 2018) and abandonment (Thompson 2021) due to a drone activity. During a fieldtrip to Lipson Island, we observed a local flying their drone around the island and landed it onto the island as it ran out of battery, this observation did not occur while the terns were breeding. Although, due to the sensitivity of tern species and the disturbance already experienced from our surveys earlier in the day, if a drone was used to fly over the colony, or possibly land near the colony, this potentially could have provoked nest abandonment. Unfortunately, the Lipson Island colony of crested terns had a failed breeding season 2021/22, which can further halt the growth on this colony and with increased human activity from the operation of a port facility and increased attention brought to Lipson Island, this has the potential to deter the colony from breeding in future years.

3.4 Conclusion

Overall, our study on the behavioural and physiological impacts of construction noise on incubating little penguins found a significant increase in vigilance behaviour but no impact to heart rate or breeding success of the colony. As there is limited literature available that illustrates how seabirds' response to introduced noise, disrupting their sensory environments, produced for coastal developments it is important to conduct further research in this area as seabird's species are known to vary in response and sensitivity to sensory disturbance (Ainley et al. 2001; Melaine 2004; Rodrigues et al. 2012, 2017a; Bevan et al. 2018; Thompson 2021; Dann & Chambers, 2013, Buxton et al., 2017; Gineste et al. 2017; Franks 2017). Determining the true impact of a new stimuli on a seabird colony can be difficult as they are faced with a complex array of pressures but future studies looking to investigate the degree of impact anthropogenic noise are recommended to examine previously undisturbed colonies to record the initial response to an introduced stimulus.

3.5 Implications of Port Spencer

The construction of a new grain export port facility, Port Spencer, has been scheduled to begin in June 2022 (Peninsula Ports 2022). The new port facility will bring about increased human presence, traffic (both terrestrial and marine), pollution such as grain, debris, light and noise, and potentially leading to the attraction of more predator and invasive species within the area (Thayer et al. 1999; Rodríguez et al. 2015, 2017a; Buxton et al. 217a; Franks 2017; PlanSA 2020; Syposz et al. 2021). The breeding seabird colonies of Lipson Island have not been monitored for a minimal of two consecutive years (Australia 2020), and no official population trends can be sourced for the little penguins, black-faced cormorants and crested terns. Therefore, monitoring the impacts from the construction and operation of Port Spencer may prove difficult due to the inability to reliably differentiate whether population decline is due to the coastal development or previous underlying conditions. From the results collected in this study, little penguins showed a significant increase in vigilance behaviours when exposed to construction noises, as a significant result came about in our short-term study, we predicted that disturbance to the colony would be exacerbated with coastal development construction and operations. The low reproductive output of black-faced cormorants and crested terns suggests these colonies are already under-pressure from the current levels of human activity within the area and without appropriate monitoring and implementation of mitigation strategies that ensure reduced human disturbance, these colonies may severely decline to a point of species extirpation within the area. Therefore, we recommended active and continuous monitoring of the seabird colonies of Lipson Island to determine the level of impact caused by human disturbance on the island, especially during the summers months when human activity is at its highest. As the port will be constructed it is important to ensure mitigation for other human activities, such as no public access on the island or no dogs or driving allowed on the mainland beach, to levitate some of the pressure that will be put on these colonies from the establishment of this coastal development 1.5 km away from the island.

Table 1 Outputs from the generalised linear mixed models and the general linear models assessing the behavioural and physiological responses of little penguins to the playback of construciton and rainfall noises.

Response Variable	**Fixed factors**	**Estimate**	**Standard Error**	***t***	***p***
Behavioural recovery from deployment (n = 22)	Intercept	730.93	166.13	4.40	< 0.001
	Sex	-64.93	109.48	-0.59	0.56
	Egg Acceptance	378.79	330.24	1.15	0.27
Heart rate recovery from deployment (n = 12)	Intercept	1529.19	343.74	4.45	0.00
	Sex	191.62	360.1	0.53	0.61
	Time sat on egg	0.25	0.35	0.70	0.50
Latency (n = 22)	Intercept	-0.29	1.09	-0.26	0.79
	Sex	1.52	1.15	1.32	0.20
	Playback Type	1.45	1.06	1.37	0.18
	Playback Order	1.26	1.06	1.20	0.24
Vigilance Intensity (n = 22)	Intercept	-16.01	23.88	-0.67	0.51
	Sex	40.80	24.01	3.78	**< 0.001**
	Playback Type	61.10	16.15	0.23	0.82
	Playback Order	3.66	16.07	1.70	0.10
Distress Intensity (n = 22)	Intercept	2.73	1.40	1.95	0.06
	Sex	1.67	1.06	1.57	0.13
	Playback Type	-1.77	1.08	-1.64	0.11
	Playback Order	-1.96	1.08	-1.82	0.08
Heart rate Intensity (n = 15)	Intercept	-1.87	5.09	-0.37	0.72
	Sex	2.07	4.75	0.44	0.67
	Playback Type	7.86	4.72	1.67	0.11
	Playback Order	-3.34	4.66	-0.72	0.48
Behaviour Recovery from Playback (n = 22)	Intercept	143.51	196.76	0.73	0.47
	Sex	-152.30	198.85	-0.77	0.45
	Playback Type	324.86	131.96	2.46	**0.02**
	Playback Order	339.27	131.25	2.59	**0.01**
Heart rate recovery from Playback (n = 15)	Intercept	2.03	1.21	1.68	0.11
	Sex	-1.14	1.00	-1.14	0.27
	Playback Type	-0.92	0.97	-0.95	0.35
	Playback Order	-0.35	0.93	-0.38	0.71
Breeding success (n =66)	Breeding Success	-1.84	0.89	-2.06	0.05
	Nest tested	1.99	0.72	2.74	**0.01**
	No. of tested nest within 15 m	0.02	0.04	0.43	0.67

Figure 1 Maps of the location of the study area within Australia (A), the location of Lipson Island (study site) in relation to Port Spencer within the Lipson Cove area (B), and the location of each species' breeding site within Lipson Island (C).

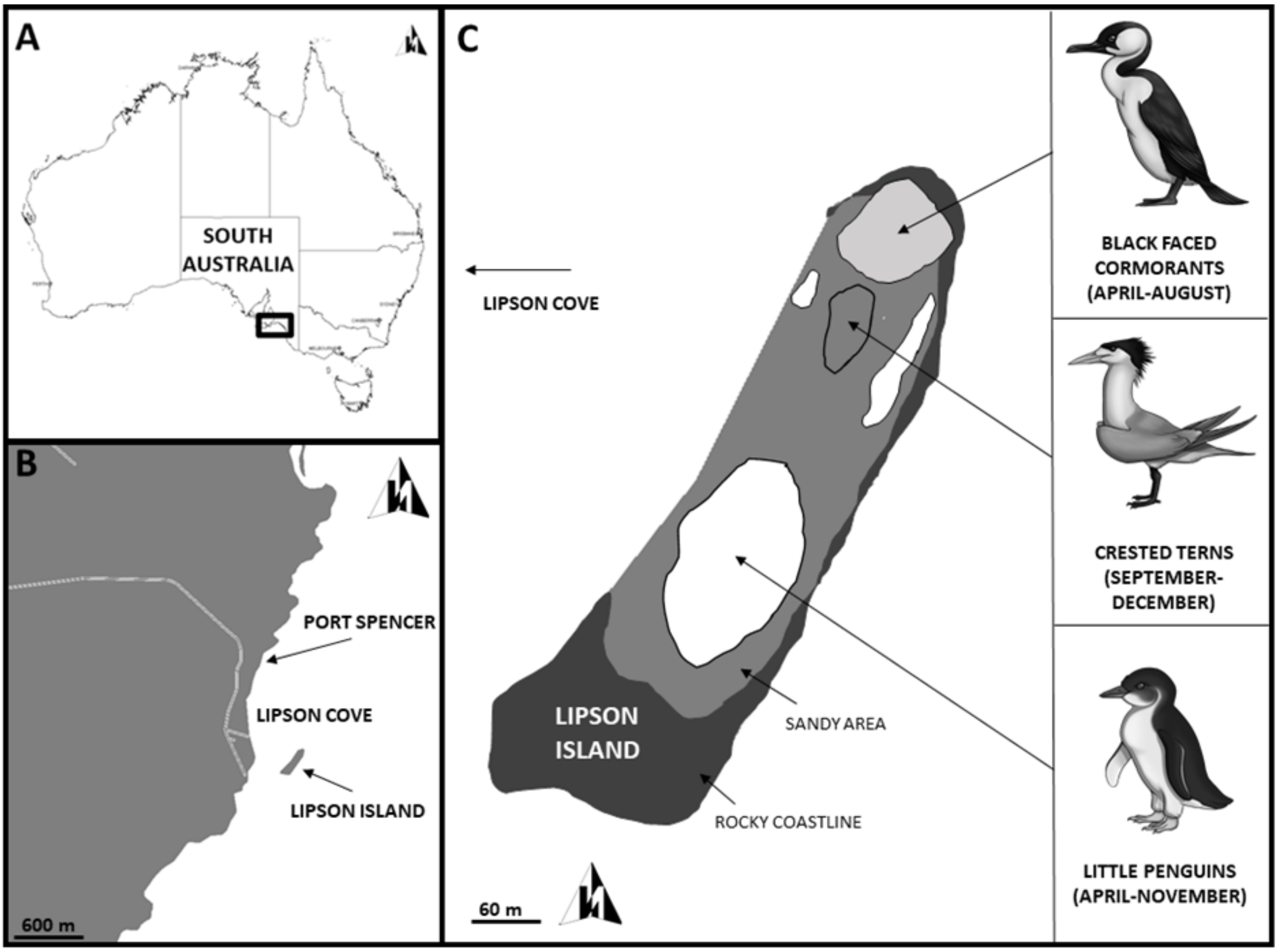

Figure 2 Response (Vigilance Intensity, calculated as baseline vigilance - playback vigilance) of little penguins (n = 22) during the playback of construction and rainfall noises on Lipson Island. Horizontal lines within the boxes represent the means. The upper and lower limits of the boxes show the 75th and 25th percentiles, respectively. Black circles indicate outliers.

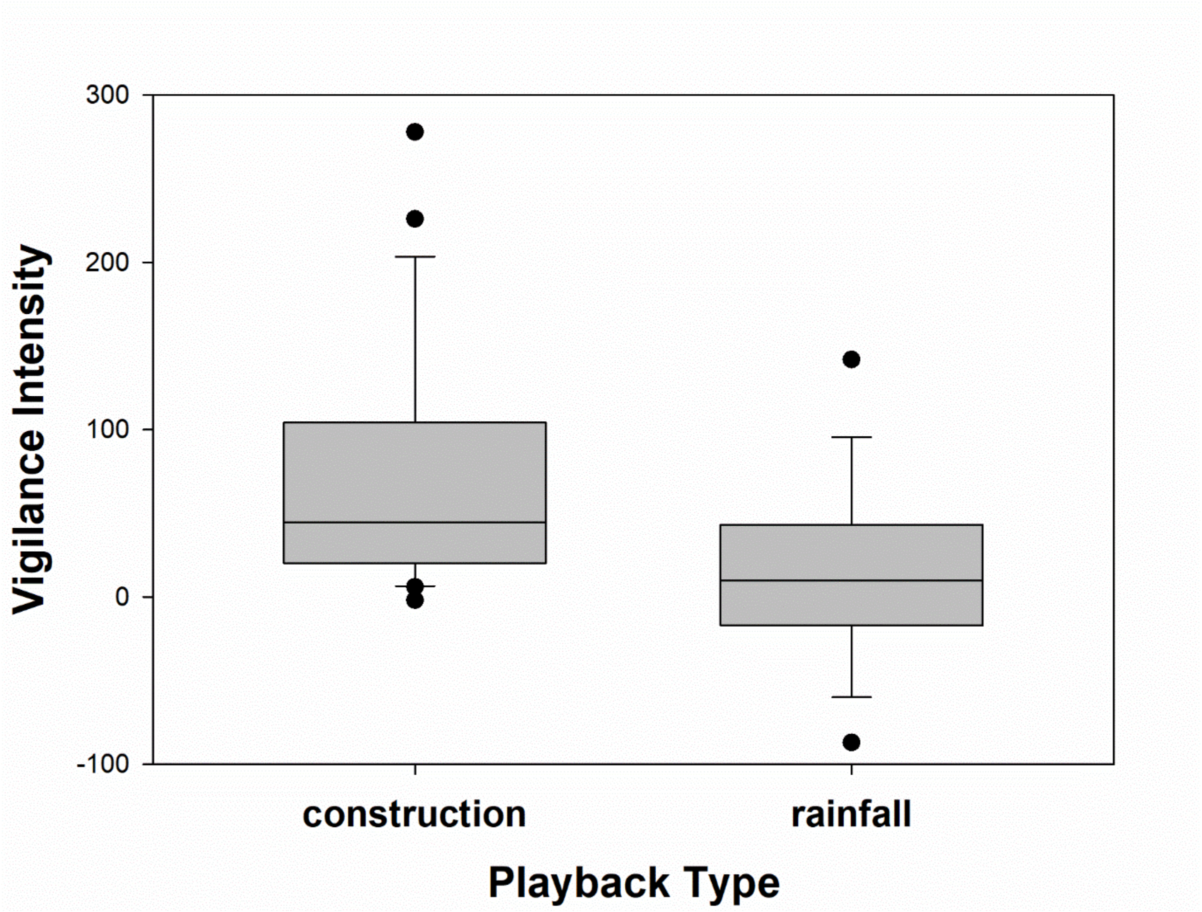

Figure 3 Behavioural recovery from playback (in seconds) of little penguins (n = 22) during the playback of construction and rainfall noises on Lipson Island. Horizontal lines within the boxes represent the means. The upper and lower limits of the boxes show the 75th and 25th percentiles, respectively. Black circles indicate outliers.

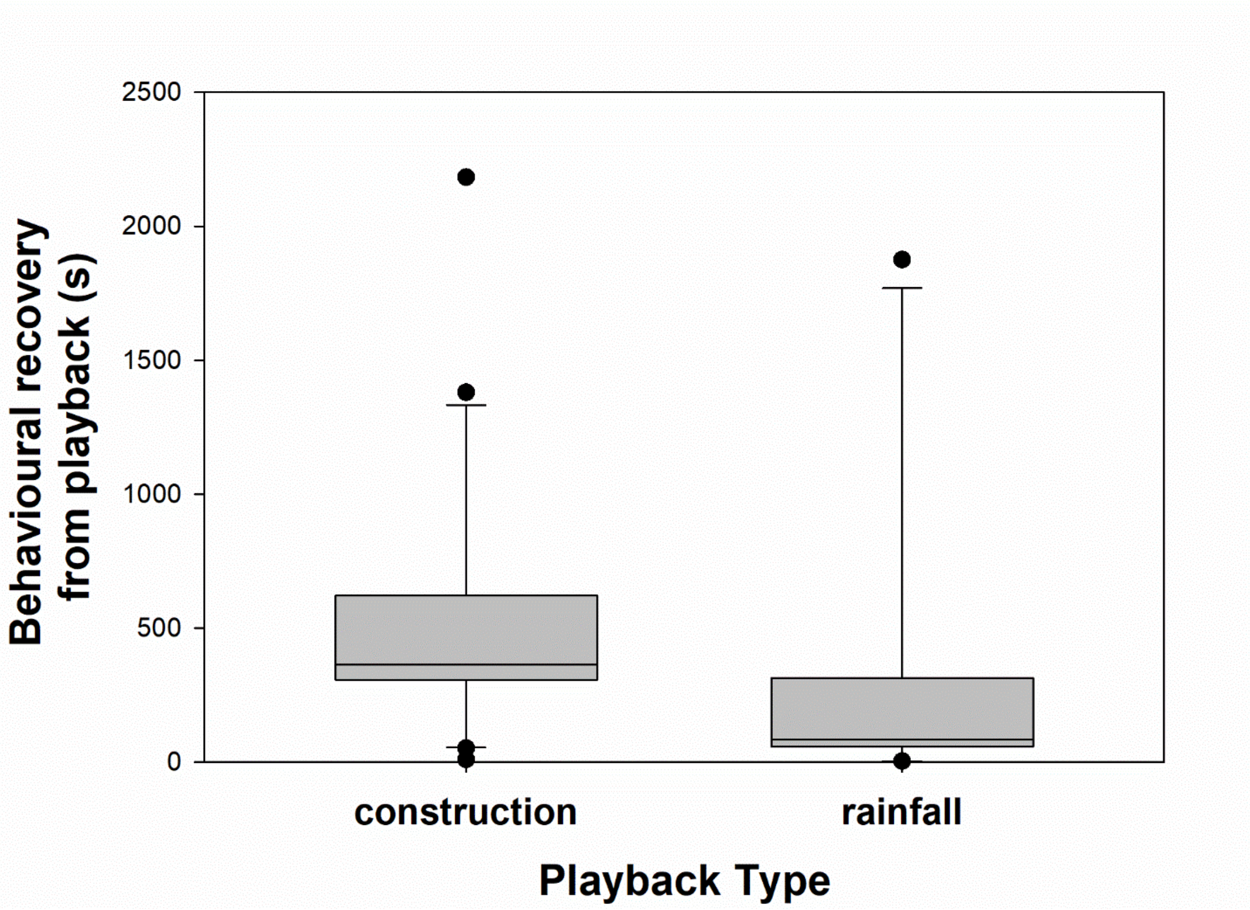

Figure 4. Breeding success of little penguins (n = 66) during the playback of construction and rainfall noises on Lipson Island.

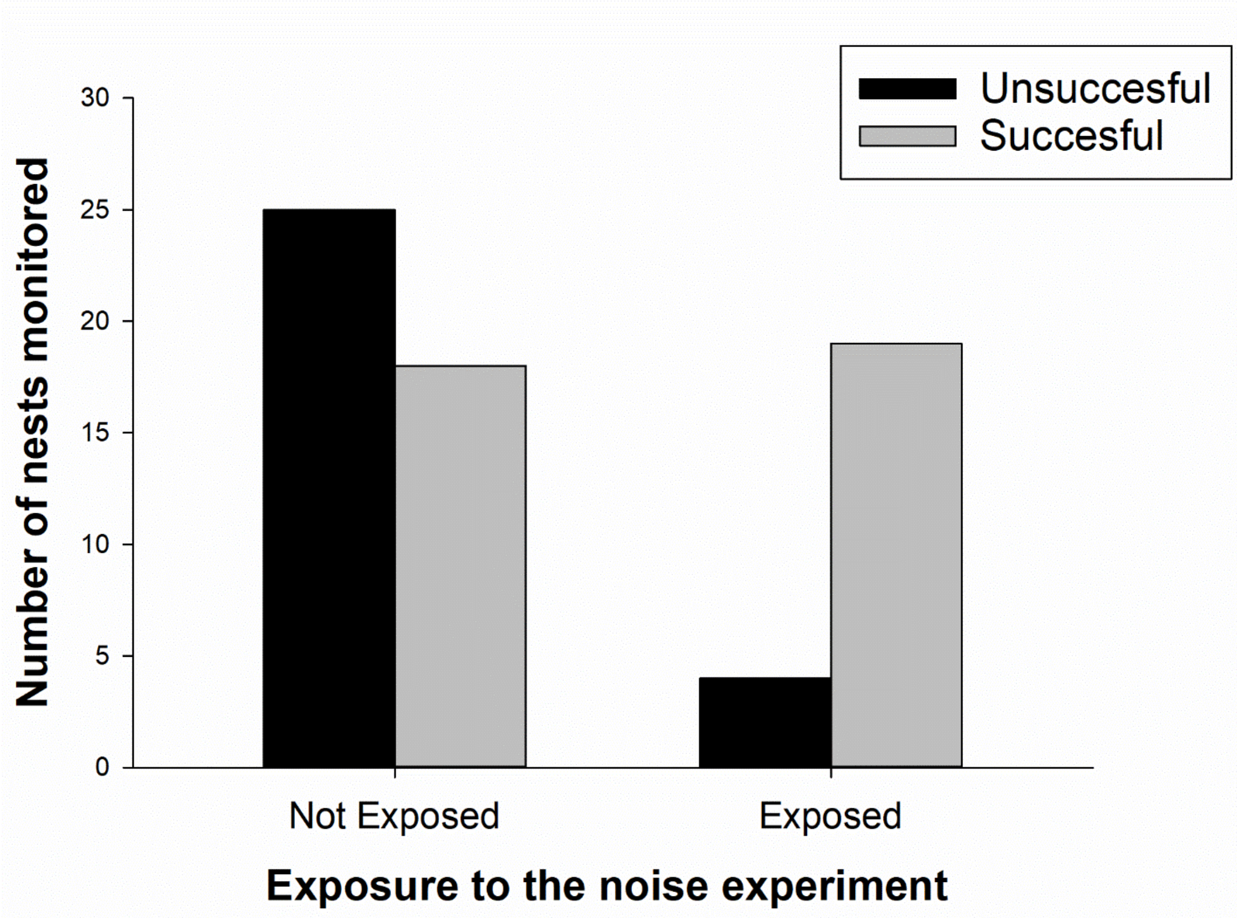

Figure 5 Breeding seabirds on Lipson Island. (A) adult little penguin sitting on eggs. (B) Two little penguin chicks 2-3 weeks old. (C) nesting black-faced cormorants on the north tip of island, in April 2021. (D) Black-faced cormorant chicks fledged from nests and aggregating along coastline, on north tip of island, in May 2021. (E) Crested terns incubating eggs in nesting site, on central northern end of island late October. (F) Crested tern eggshell remnants.

CHAPTER 4 – BRIGHT AND EARLY: ARTIFICIAL LIGHT AFFECTS RETURN TIME IN LITTLE PENGUINS

4.1 Introduction

Humans utilise artificial light at night (ALAN) for safety, amenity and increased productivity, but its presence in the environment can interrupt the natural day-night regime of wildlife, alter behavioural time budgets and change the distribution of species within and around an illuminated area (Kempenaers et al. 2010; Rodríguez et al. 2016; Kyba et al. 2017). Silva et al. (2017) found that, in artificially illuminated nesting grounds, female loggerhead turtles (*Caretta caretta*) had reduced nesting attempts, took more time to construct nests, and took longer to depart from the nesting site. The authors also examined the response of ghost crabs (*Ocypode cursor*), a predator of loggerhead turtle nestlings, and found that ghost crab presence increased under illuminated conditions, increasing predation risks for turtle nestlings (Silva et al. 2017). Therefore, not only did artificial light alter the behavioural time budget of individuals trying to complete biologically important tasks, such as breeding, but it also increased the foraging efficiency of their predators (Silva et al. 2017; Dominoni et al. 2022), all of which can lead to cascading impacts throughout the trophic levels resulting in reduced biodiversity and species abundance (Bennie et al. 2018). Therefore, understanding the altered dynamics between and within species under ALAN is important to identify the most vulnerable species and create focused mitigation strategies.

Seabirds are highly sensitive to the violet-blue region of the visible spectrum (380 – 440 nm) and with light-emitting diode (LED) white lights becoming popular for outdoor lighting, as they are cost-efficient and produce higher radiance and thus increased disturbance to surrounding wildlife (Morgan-Pattison et al. 2018; McNaughton et al. 2021; Australia 2020). Due to the sensitivity of seabirds to short-wavelength individuals can be attracted illuminated areas as they may use the presence of light as visual cues (Wiese et al. 2001) or may have associated artificial light with prey location (e.g., fishing boats using flood lights attracts foraging seabirds; Merkel & Johansen 2011). Artificial light at night (ALAN) has been found to confuse and distract nocturnal seabird species (Rodríguez et al. 2017a) and attract individuals to illuminated areas exposing them to risks of predation or collisions with vehicles, structures, or surfaces (Wiese et al. 2001; Merkel & Johansen 2011; Deppe et al. 2017; Rodríguez et al. 2017a).

The effects from artificial light can be exacerbated by environmental factors such as moon phase, cloud cover, precipitation and proximity to nesting site (Wiese et al. 2001; Chiaradia 2007; Australia 2020) increasing mass grounding events (Ainley et al. 2001; Raine et al. 2007; Merkel & Johansen 2011; Rodríguez et al. 2012, 2017a; Gineste et al. 2017; Deppe et al. 2017) with fledglings being the most vulnerable to the adverse impacts as their naivety attracts them to illuminated areas and into dangerous environments (Ainley et al. 2001; Rodríguez et al. 2017a).

Yet there is some conflicting research on the impact of ALAN on seabirds as Rodríguez et al. (2018) found that little penguins (*Eudyptula minor*) returning to their nests at night on Phillip Island (Victoria) preferred artificially illuminated paths as it increased their visibility (due to improved navigation and predator detection). However, these results may have been influenced by the fact that the Phillip Island colony have used artificial conditions for more than 100 years (Rodríguez et al. 2018), and the penguins may have become somewhat habituated to artificial lighting.

Most studies investigating the impact of human disturbance on seabird colonies assess breeding success to estimate any potential impact of the disturbance but rarely monitor colony attendance of individuals under disturbed conditions. While natural variation in colony attendance by breeding seabirds can depend on temporal (e.g., stage of breeding) (Wails et al. 2020) and environmental factors (e.g., weather conditions or prey availability) (Harding et al. 2007; Henson et al. 2007), increased human disturbance can also lead to reduced colony attendance (Thayer et al. 1999; Buxton et al. 2017a). During the breeding season, seabirds have a reproductive obligation to return to their colonies from foraging trips, with the highest colony attendance occurring during pre-egg laying and incubation periods (Stahl & Sagar 2006; Harding et al. 2007; Wails et al. 2020). If adults do not attend to their offspring on a regular basis, this could lead to reduced chick body mass or mortality from starvation or predation (Anderson & Keith 1980; Cianchetti-Benedetti et al. 2018). Disturbance of breeding seabirds during these highly-energy demanding periods can also cause momentary or permanent nest abandonment due to avoidance behaviours (Anderson & Keith 1980). Thayer et al. (1999) monitored the colony attendance of several breeding populations of common murres (*Uria aalge*) following a recent shipwreck and showed that birds significantly reduced their attendance to the colony due to the disturbance and consequent increased presence of avian predators (Thayer et al. 1999). Therefore, assessing whether colony attendance may be altered as a result of human disturbance is important to determine whether parental obligations are being interrupted.

Unlike humans, animals only exhibit behaviours, when necessary, as fitness pressures and energy requirements mean every behaviour is significant and altered behavioural time budgets can lead to wasted energy reserves or reduced fitness (Anderson & Keith 1980; Burger 1998; Ronconi & Clair 2002; Owens et al. 2020; Dominoni et al. 2022). Seabirds are intuitive animals that rely on their senses to perform daily tasks (Friesen et al 2017; Rajpar 2018) but can lead to their behavioural time budget being easily swayed by sensory disturbance produced from anthropogenic activities. Human disturbance often causes individuals to display increased vigilance and avoidance behaviours (Yorio & Boersma 1992; Buxton et al. 2017; Colombelli-Négrel & Katsis 2021). These stress responses can consume precious energy reserves while also diverting attention away from biologically important tasks (Derose-Wilson et al. 2015; Franks 2017; Bevan et al. 2018) such as foraging (Lebbin et al. 2007) and breeding (Cianchetti-Benedetti et al. 2018). Therefore, it is important to understand how a human disturbance can impact behavioural time budgets of seabirds to ensure vulnerable colonies, already exposure to natural pressures (such as predation or prey availability), are not experiencing unnecessary energy expenditure, especially during high-energy demanding periods such as breeding.

South Australia is believed to host approx. 100 Little penguin colonies, with 20+ colonies suspected to have declined in the last decades for reasons not well understood (Weibkin 2011; DEWNR 2016; Colombelli-Négrel 2017). Little penguins are listed as least concern due to their large distribution range across southern Australia and New Zealand, but they have experienced localised declines across their distribution (Weibkin 2011; McLeay et al. 2009b). There has been little research done that investigates the response of little penguins have to ALAN (Rodríguez et al. 2018) and no study has investigated the behavioural time budget of returning little penguins under artificially light conditions. For populations unfamiliar with ALAN and unprotected from conservation groups it is important to understand their response to ALAN to determine whether conservation strategies are needed for these less protected colonies to prevent population declines. As South Australian colonies of little penguins are of conservation concern, and due to the small size of the Lipson Island colony, it is important monitor each population that is threatened by the introduction of coastal developments as smaller populations are more vulnerable to change then larger populations (Richards et al. 2021). We want to determine whether the presence of LED white light will alter the behavioural time budgets of foraging little penguins returning to their breeding site at night. The need to investigate the impacts of ALAN on this colony of little penguins is due to a proposed port facility, Port Spencer, planned to be built 1.5 km away from Lipson Island. In 2020 a government document was released that suggests that seabird colonies in the impact zone for light pollution should have a minimum of two years monitoring of both adults and fledglings prior to development construction (Australia, 2020) but in the case of Port Spencer, their Public Environment Report only surveyed the seabirds of Lipson Island over a two-day period back in 2011 (Madden-Hallet et al. 2011). Therefore, not only is there a need for a more reliable breeding monitoring program but also to investigate the impact of ALAN prior to the construction of the port. In this study, we investigated whether the presence of artificial light on Lipson Island (South Australia) predicted little penguins' behaviours and attendance at night. We predicted that little penguins would arrive significantly earlier under artificially illuminated conditions, due to increased visibility (Rodríguez et al. 2018). We also predicted that vigilance behaviours would be less often observed but that breeding behaviours would be more often observed under artificially illuminated conditions (Rodríguez et al. 2018).

4.2 Methods

4.2.1 Study location

We conducted our light experiment on Lipson Island Conservation Park (34°26'S, 136°26'E; 2021) located in Lipson Cove (figure 1) 70 km north of Port Lincoln, South Australia, between May and

October 2021. The island is 336 m long with a maximum width of 100 m and is located 250 m from the mainland beach of Lipson Cove (Figure 6). The water depths off the east coast of the island drop to a depth of 15 m within 200 m of Lipson Island, while the water depths on the north-west side are approximately 5 m (Charts, 2022). The adjacent mainland coast is largely agricultural with a small campsite located at Lipson cove. Therefore, the area has no permanent artificial light sources, but short term artificial light presence can come from campers' flashlights or boats passing by at night.

4.2.2 Study species

Little penguins breed between April and March (Reilly & Cullen 1981; Johannesen et al. 2003; Colombelli-Négrel 2015). Little penguins have two peak breeding periods: one in autumn by older, more experienced penguins and a second peak in spring/summer when older individuals attempt their second round of breeding and younger individuals undertake their first round (Ramírez et al. 2020). During incubation and the first two weeks of chick-rearing, the male and female take turns; while one is out foraging, the other stays at the nest incubating the eggs or guarding the chicks (Miyazaki & Waas 2003). Incubation period lasts for 33-44 days, and chicks fledge 8-9 weeks after hatching (Kemp & Dann 2001; Colombelli-Négrel 2015). Chick-rearing occupies 16 % of the annual time budget of an adult little penguin and consumes 31 % of their annual energy time budget as adults need to consume 60 % of their body weight in prey daily (Gales & Green 1990). Little penguins travel up to 20 km from their colony in one day while provisioning for their chicks (Collins et al. 1999). Little penguins are highly dependent on their vision in order to catch prey and therefore can only effectively forage when there is an abundance of light (Cannell & Cullen 1998). Therefore, they forage only during daylight (Cannell & Cullen 1998; Collins et al. 1999), returning ashore at sunset (Klomp & Wooller 1991). Arrival times of little penguins to their breeding site can vary depending on prey availability and distribution (i.e., low prey supply in proximity to breeding site may force individuals to travel further away from the colony), extreme weather conditions (i.e., strong winds and turbulent seas can disorientate individuals), or because of other unforeseen barriers (Chiaradia 2007). Little penguins generally return to their nests within 60 minutes of sunset (Daniel et al. 2007), although one study found that penguins continue to forage up to 26 minutes after sunset (Wiebkin 2012) and another found that, in years of low breeding success, little penguins tend to arrive on average 30 min later than expected (Chiaradia 2007).They are known to form groups of 5 – 10 individuals when returning to shore (Daniel et al. 2007) and tend to cross landing sites in darkness to avoid predation from diurnal birds of prey or pacific gulls (*Larus pacificus*), however, gulls are known to continue to hunt at night and therefore, penguins should remain cautious when crossing (Chiaradia 2007).

4.2.3 Experimental set up

To determine the best locations for our experiment, we first installed four infrared motion sensor cameras (Argus 2, Reolink, Australia) installed approximately 30cm above ground level at various locations surrounding the little penguins nesting area for one month in May 2021. We programmed the cameras to detect movements continuously for four hours starting 30 minutes before sunset, and all cameras were set to record on the longest setting once triggered (30s; however, once trigged and if continuous movement(s) occurred, the cameras could record for up to 5 minutes continuously). In June 2021, to experimentally test little penguins response to artificial light, we installed (1) a self-sustaining light system, composed of one white LED floodlight (DIY LED Portable Floodlight, 15W 220-240V ~50Hz; ARLEC, Australia; light colour 5700K) connected to an inverter (300 watt, Intelli-Wave Pure Sine) linked to a deep cycle battery (100ah AGM, Predator) automatically charged by one Solar Panel (100 Watt, Mono), and (2) a control system with a decoy light (turned off) that mimicked the experimental system at two different sites (site 1, site 2) separated by 100m (Figure 7). Each system was paired with two infrared motion sensor cameras, installed approximately 30cm above ground level and facing landing locations and crossing zones used by the penguins at night (determined from our initial recordings in May), to record little penguins' behaviours and arrival times. We programmed the light to turn on 30 minutes before sunset and to stay on for two hours and the cameras to detect movements continuously for four hours starting 30 minutes before sunset. All cameras were set as previously described. To further test whether behaviours were influenced by our experimental set up or simply due to the location within the colony, we alternated the location of the experimental and control systems between the two sites (sites 1 and 2) every three weeks (Figure 7).

From the video recordings, we recorded the following behaviours: (1) 'time of arrival' (sighting of the first individual on the rocky coastline arriving from the water); (2) 'group size' (number of individuals within the same video recording on the rocky coastline arriving from the water); (3) 'vigilance behaviour' (whether individuals within a video engaged in vigilance behaviour; yes/no – we recorded vigilance as occurring if one or several individual(s) scanned the environment, generally with its/their neck extended and its/their head upright moving quickly from right to left, and displayed intense staring; Fowler 1999; Holmes et al. 2005; Sherwen et al. 2015; Schaefer and Colombelli-Négrel 2021); (4) 'breeding behaviour' (whether individuals within a video engaged in breeding behaviours; yes/no – we recorded breeding behaviours as occurring if one or several individual(s) displayed mating, courting or mate/nest guarding behaviours, or if it/they were searching for nesting material).

To determine if other factors may have influenced little penguins' arrival times and behaviours, we also recorded several environmental factors known to influence seabirds behaviours: (1) 'moon illumination' (percentages of illumination, obtained from the following website: https://www.almanac.com/astronomy/moon/calendar and categorised as 0-25%, 26-50%, 51-75% or 76-100%) as seabird nocturnal activity significantly increases during nights with high illumination (Cannell & Cullen 1998; Wiese et al. 2001; Rodríguez et al. 2018), and (2) 'wind speeds' [recorded from the Port Lincoln weather station from http://www.bom.gov.au/climate and categorised as none to light wind (<11 km/h), moderate wind (<29 km/h), strong wind (<50 km/h), gale force wind (<75 km/h)] as wind speeds have been identified to influence the behaviours and vocalizations of seabirds, including little penguins (e.g., Dehnhard et al. 2013, Pistorius et al. 2015, Saraux et al. 2016).

4.2.4 Data analysis

We used SPSS version 25.0 for Windows (SPSS Inc., Chicago, IL, U.S.A) for all statistical analyses. Data are shown as mean ± standard deviation unless otherwise stated. Prior to analysis, we assessed collinearity between (1) continuous predictors using the variance inflation factors (VIF) analysis or (2) categorical predictors using Pearson's correlations. In all models, VIF values were well < 2, and Pearson's coefficients were close to zero confirming no collinearity (Zuur et al. 2009; Fox et al. 2015). We categized little penguin 'arrival times' as either: 'on-time' (*arrived within 60 minutes of sunset)* or 'late' (*arrived more than 60 minutes after sunset*) and examined 'arrival time' using a Generalized Linear Model (GLM) analysis with a binomial distribution and a logit link function with 'light' (*on, off*), 'site' (*site 1, site 2*), 'light*site', 'wind speeds', 'moon illumination' and 'group size' as fixed factors. We analysed 'vigilance behaviour' and 'breeding behaviour' using GLMs with a binomial distribution and a logit link function with 'light', 'site', 'light*site', 'no. penguins' (*number of individuals present in the video clip*), 'wind speeds' and 'moon illumination' as fixed factors.

4.3 Results

The number of clips obtained by the four cameras each night varied between two and 108 clips, with 11,404 video clips recorded over 133 nights. As the experimental system did not work on majority of the nights due to insignificant sunlight to recharge the battery, we only used footage from 47 nights for the analyses

4.3.1 Arrival time

Little penguins arrived to the landing site significantly more often late when the light was on compared to when the light was off (GLM: 'light', $F_{1,84} = 10.66$, $p = 0.002$; Table 2, Figure 8). None of the remaining factors were correlated to 'arrival time' (GLM: 'group size' $F_{1,84} = 0.12$, $p = 0.74$; 'wind speed' $F_{1,84} = 1.58$, $p = 0.19$; 'moon illumination' $F_{1,84} = 0.86$, $p = 0.47$; 'site' $F_{1,84} = 3.68$, $p = 0.06$; 'light*site' $F_{1,84} = 0.65$, $p = 0.42$; Table 2).

4.3.2 Vigilance behaviour

Little penguins that arrived in site 2 significantly displayed more vigilance behaviours compared to individuals in site 1, with individuals in site 2 displaying more vigilance when the light was on compared to individuals in site 1 that showed less vigilance when the light was on (GLM: 'site' $F_{1,68} = 5.39$, $p = 0.02$; 'light*site $F_{1,68} = 5.77$, $p = 0.02$; Table 2, Figure 9). None of the remaining factors were correlated to 'vigilance behaviour' (GLM: 'penguins in view' $F_{1,68} = 0.43$, $p = 0.52$; 'wind speed' $F_{1,68} = 0.67$, $p = 0.62$; 'moon illumination' $F_{1,68} = 2.20$, $p = 0.10$; 'light' $F_{1,68} = 0.34$, $p = 0.56$; Table 2).

4.3.3 Breeding behaviour

Mean of 3 individuals were present when breeding behaviours occurred with a significant correlation to conspecific presence (GLM: 'penguins in view' $F_{1,47} = 6.48$, $p = 0.01$; Table 2, figure 10). None of the remaining factors were correlated to 'breeding behaviour' (GLM: 'light' $F_{1,47} = 0.96$, $p = 0.0.33$; 'wind speed' $F_{1,47} = 0.25$, $p = 0.91$; 'moon illumination' $F_{1,47} = 0.94$, $p = 0.43$; 'site' $F_{1,47} = 0.75$, $p = 0.39$; 'light*site' $F_{1,47} = 0.15$, $p = 0.70$; Table 2).

4.4 Discussion

Colony attendance by breeding seabirds is important due to the parental care provided to off-spring which if altered can lead to reduced breeding success (Thayer et al. 1999; Buxton et al. 2017a). In this study, we found that little penguins arrived later to their landing site on nights when the artificial light had been turned on. We found that little penguins arriving in site 2 significantly displayed more vigilance behaviours compared to individuals in site 1, with individuals in site 2 displaying more vigilance when the light was on compared to individuals in site 1 that showed less vigilance when the light was on. We also found, that breeding behaviour occurrence was significantly correlated to conspecific presence. These results indicate that little penguins on Lipson Island, with no previous history of artificial light, arrived later with behavioural differences between individuals arriving in site 1 compared to site 2.

Little penguins arriving to the landing site on nights when the light was on did not increase, i.e., did not attract individuals to the landing site, and individuals tended to arrive later on nights when the light was on. Some considerations to be aware of, when observing footage of arrival times, during dark nights our vision was restricted and therefore arrival times may not be accurate, whereas when the light was on, we could clearly see when a penguin arrived. Another factor that could reduce the reliability of recorded arrival times is the terrain of the coastal, as it was rocky, penguins arriving may have lingered behind the rocks, waiting for conspecifics (Daniel et al. 2007), before crossing into visible range. Little penguins are known aggregate offshore before arriving on land and they have been recorded to remain at the landing site for up to 2 hours before crossing (Chiaradia 2007). Previous studies have documented little penguins to be undeterred from returning to their breeding colony in the presence of artificial light (Rodrigues et al. 2016, 2018). Rodrigues et al. (2018), found that light intensity did not impact little penguin colony attendance and that returning little penguins preferred well-illuminated tunnels or paths over those poorly illuminated. Though these studies occurred on Phillip Island (Victoria) an area with more than 100 years of artificial light disturbance. Our study has found contradicting results as little penguins tended to arrive on shore after the light had turned off. As our study has limited data available from nights with the light on we recommend future studies collecting a larger sample size of nights with the light on and also recording arrival times throughout a larger portion of the night to determine whether permanent presence of artificial light at night will completely deter returning individuals. Overall, we were able to determine a significant decrease in ‘on time’ arrivals of little penguins exposed to artificial light on Lipson Island but more research is needed to determine whether arrival frequencies may also be impacted by artificial light presence. A lot of research has been conducted on flying-seabird species, such as petrels, monitoring the impacts of artificial light and nocturnal activity (Ainley et al. 2001; Wiese et al. 2001; Merkel & Johansen 2011; Deppe et al. 2017; Rodríguez et al. 2017). But our study was the first to experimental test the impact of artificial light on a previously undisturbed colony of little penguins and has added to previous work that has investigated the arrival times of little penguins under artificially illuminated conditions (Rodrigues et al. 2016, 2018).

Other studies have also found poor weather conditions to impact colony attendance with less individuals returning to their breeding site on nights with poor visibility and that those that did arrive preferred using artificially light paths (Chiaradia, 2007). Though in our study there were nights of high winds, indicating poor weather conditions, this was not correlated to the arrival times of little penguins in our study. Rodrigues et al. (2016), found a variation between arrival times and correlations with environmental factors, such as moon illumination, suggesting site-specific differences may be a contributing factor. But in our study moon light did not significantly predict arrival times of little penguins.

Our study found no correlations between artificial light presence and altered behavioural time budgets of arriving little penguins. We documented a significant correlation between vigilance behaviour occurrence and site location, with a higher vigilance behaviour occurrence documented in site 2 compared to site 1. This may be due to the landing site at site 2 having a flat, open crossing that was more exposed, whereas the landing site at site 1 was more concaved with a shorter exposed crossing area. Little penguins have excellent vision both in and outside of the water (Coimbra et al. 2012) but reduced light may make them more cautious, i.e., vigilant, due to their vision being restricted and therefore spend more time focusing on their surroundings. Therefore, in open, dark areas little penguins may feel unsafe due to the restricted visibility and limited shelter from potential predators.

Our study did not find a correlation between breeding behaviour occurrence and artificial light presence, this may be due to little penguins mainly preforming breeding behaviour at their nest opposed to at their landing sites. Therefore, we did not find a significant correlation between little penguin behaviours and presence of artificial light which improves our understanding of how little penguin response to an altered sensory environment. As majority of previous studies investigating seabirds’ response to artificial light at night focusing on the detoured foraging activities of flying-seabird species (Hockin et al. 1992; Wiese et al. 2001; Ainley et al. 2001; Merkel & Johansen

2011; Deppe et al. 2017) this study has given insight to how non-flying penguins reaction to light presence. Future studies may seek to investigate the presence of artificial light while monitoring little penguin behaviours at their nests. This study would then be able to incorporate parental care observations under artificial light.

The presence of artificial light at night can impacted species abundance and predator-prey interactions (Dominoni et al. 2022), as well as lead to altered behaviour, fitness, and species biodiversity (Bennie et al. 2018; Owens et al. 2020). For seabird species artificial light has been largely problematic for petrel species when undertaking nocturnal foraging trips (Rodríguez et al. 2015). With previous studies founding confused and disorientated individuals being triggered to fly into structures or surfaces (Rodríguez et al. 2015). Contrary to this, little penguins have not been found to be negatively impacted by artificial light but previous research has only focus on one population. Therefore, as the reaction to artificial light can be species specific it may also by site-specific, suggesting future studies investigating the impacts of artificial light on seabird species incorporate more landscape factors that may have an effect on colony or individual behaviours.

4.4.4 Implications of Port Spencer

The presence of artificial light produced for a new port facility, Port Spencer, to be located 1.5 km away from Lipson Island is not expected to directly impact the breeding colony of little penguins but indirect impact may be possible, such as increased predator presence (Thayer et al. 1999; Rodríguez et al. 2015, 2017a; Buxton et al. 217a). The proposal for Port Spencer has stated that they will use shielded dome light for their outdoor lighting as a means to reduced unfiltered light pollution (PlanSA 2020). But no prior monitoring of seabird behaviours was conducted as part of their public environment report, with recently released guidelines, on light pollution mitigation for wildlife, recommending that breeding seabird colonies should be monitored for two consecutive breeding seasons prior to development construction (Australia 2020). In order to ensure that at important seabird rookeries and colony attendance remains constant, adults and fledglings are not grounded, and fledglings launch successfully from their breeding site (Australia 2020). As this may not be an issue for little penguins but for the flying, breeding and roosting seabirds on Lipson Island, such as the black-faced cormorants and crested terns, it could pose a substantial threat activity (Ainley et al. 2001; Wiese et al. 2001; Merkel & Johansen 2011; Deppe et al. 2017; Rodríguez et al. 2017) to colonies already showing impacts from minimal human activities (see chapter 3). Therefore, we recommend future monitoring of the seabird colonies of Lipson Island, permanent cameras set up on the island could give researchers the ability to monitor behavioural changes to night-time behaviours of breeding or roosting seabirds over a longer period of time.

Table 2 Outputs from the generalised linear mixed models assessing the behavioural response of little penguins to artificial light

Response Variable	Final Model	Coefficients	Estimate	Standard Error	*t*	*p*
Arrival Time (n = 262)	Arrival Time	Intercept	-16.2	1275.15	-0.01	0.99
	Group Size	Group size	-0.05	0.15	-0.34	0.74
	Wind	Wind	12.05	1275.15	0.01	0.99
	Moon	Moon	-0.02	0.74	-0.03	0.98
	Site	Site	-0.59	0.60	-0.98	0.33
	Light	Light	2.23	0.71	3.15	**0.002**
	Light*Site	Light*Site	-0.89	1.10	-0.81	0.42
Vigilance Behaviour Occurrence (n= 106)	Vigilance Behaviour	Intercept	-1.31	1.17	-1.12	0.27
	Penguins in View	Penguins in view	-0.07	0.11	-0.65	0.52
	Wind	Wind	0.48	1.61	0.30	0.77
	Moon	Moon	-1.21	0.65	-1.87	0.07
	Site	Site	-0.13	0.52	-0.26	0.80
	Light	Light	-1.12	0.86	-1.31	0.195
	Light*Site	Light*Site	2.88	1.20	2.40	**0.02**
Breeding Behaviour Occurrence (n = 66)	Vigilance Behaviour	Intercept	4.12	2.04	2.01	0.05
	Penguins in View	Penguins in View	-0.92	0.36	-2.55	**0.01**
	Wind	Wind	1.86	2.38	0.78	0.44
	Moon	Moon	-1.07	1.13	-0.95	0.35
	Site	Site	-0.41	1.62	-0.25	0.80
	Light	Light	-0.50	1.28	-0.39	0.70
	Light*Site	Light*Site	-0.73	1.90	-0.38	0.70

Figure 6 . Maps of the location of the study area within Australia (A), the location of Lipson Island (study site) in relation to Port Spencer within the Lipson Cove area (B), and the location of the two experimental sites (sites 1 and 2) used in this study on Lipson Island (C).

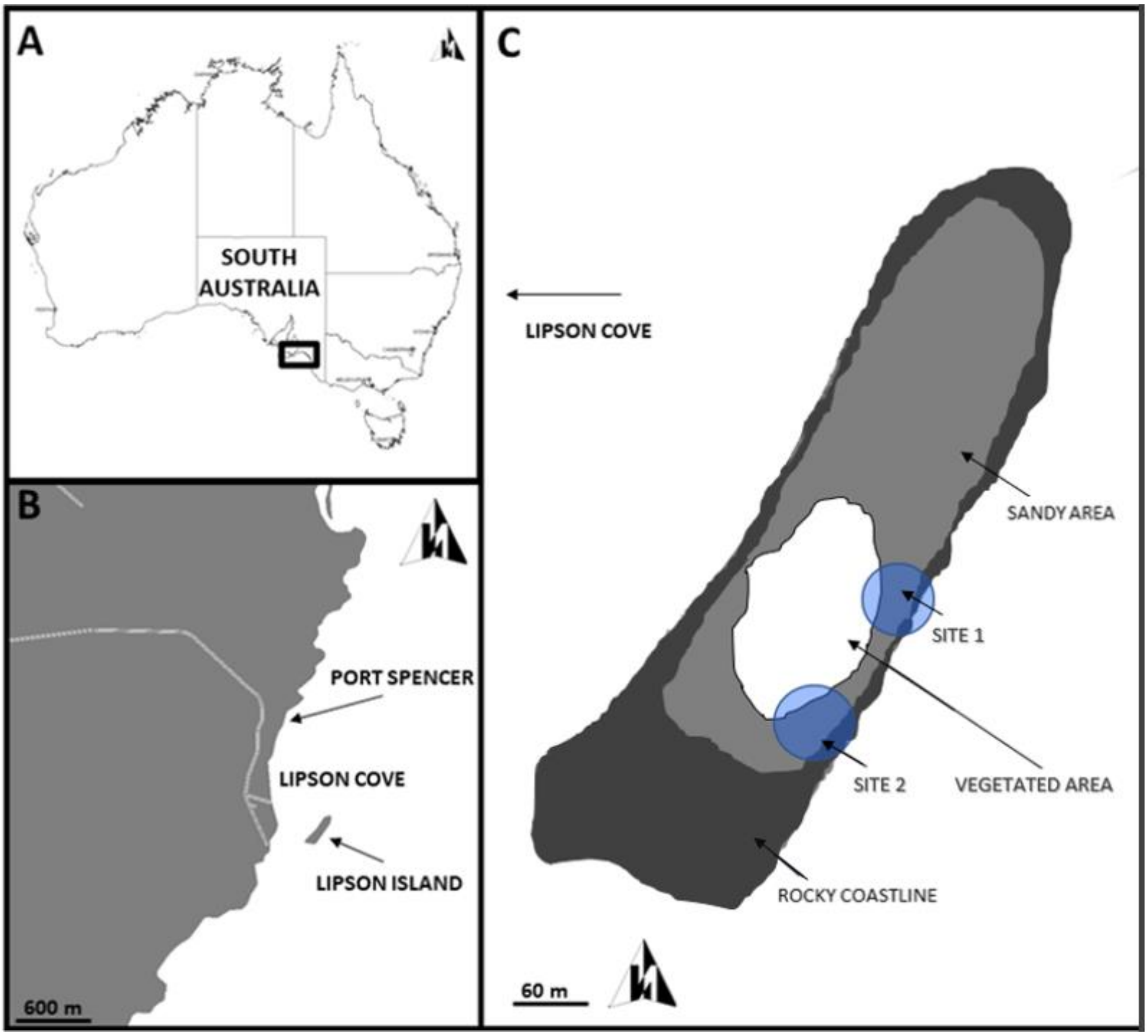

Figure 7 Experimental set up, (left) a self-sustaining light system, composed of one white LED floodlight (DIY LED Portable Floodlight, 15W 220-240V ~50Hz; ARLEC, Australia; light colour 5700K) connected to an inverter (300 watt, Intelli-Wave Pure Sine) linked to a deep cycle battery (100ah AGM, Predator) automatically charged by one Solar Panel (100 Watt, Mono), and (right) a control system with a decoy light (turned off) that mimicked the experimental system at two different sites.

Figure 8 Arrival time of little penguins during night with the light on and nights with the light off.

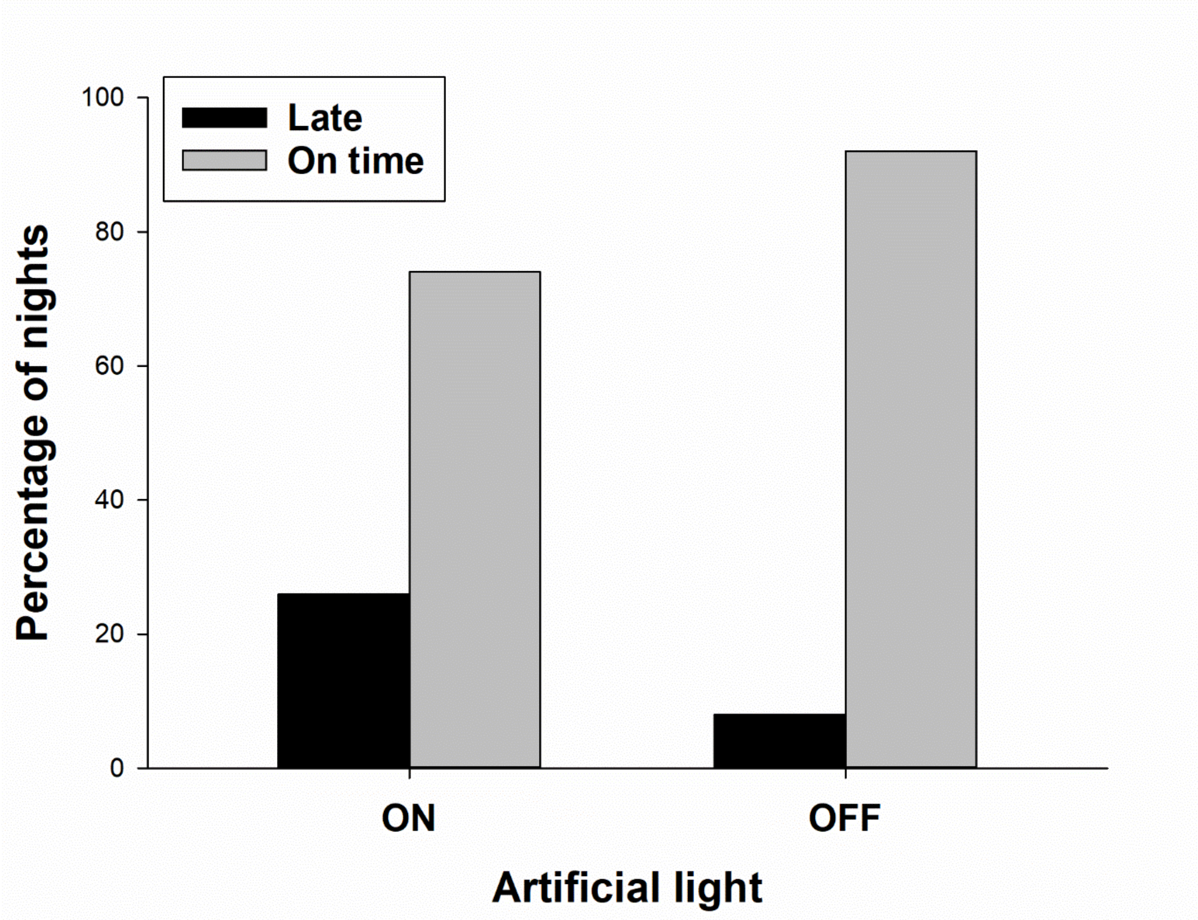

Figure 9 The occurrence of vigilance behaviours in site 1 and site 2, during night with the light on and nights with the light off.

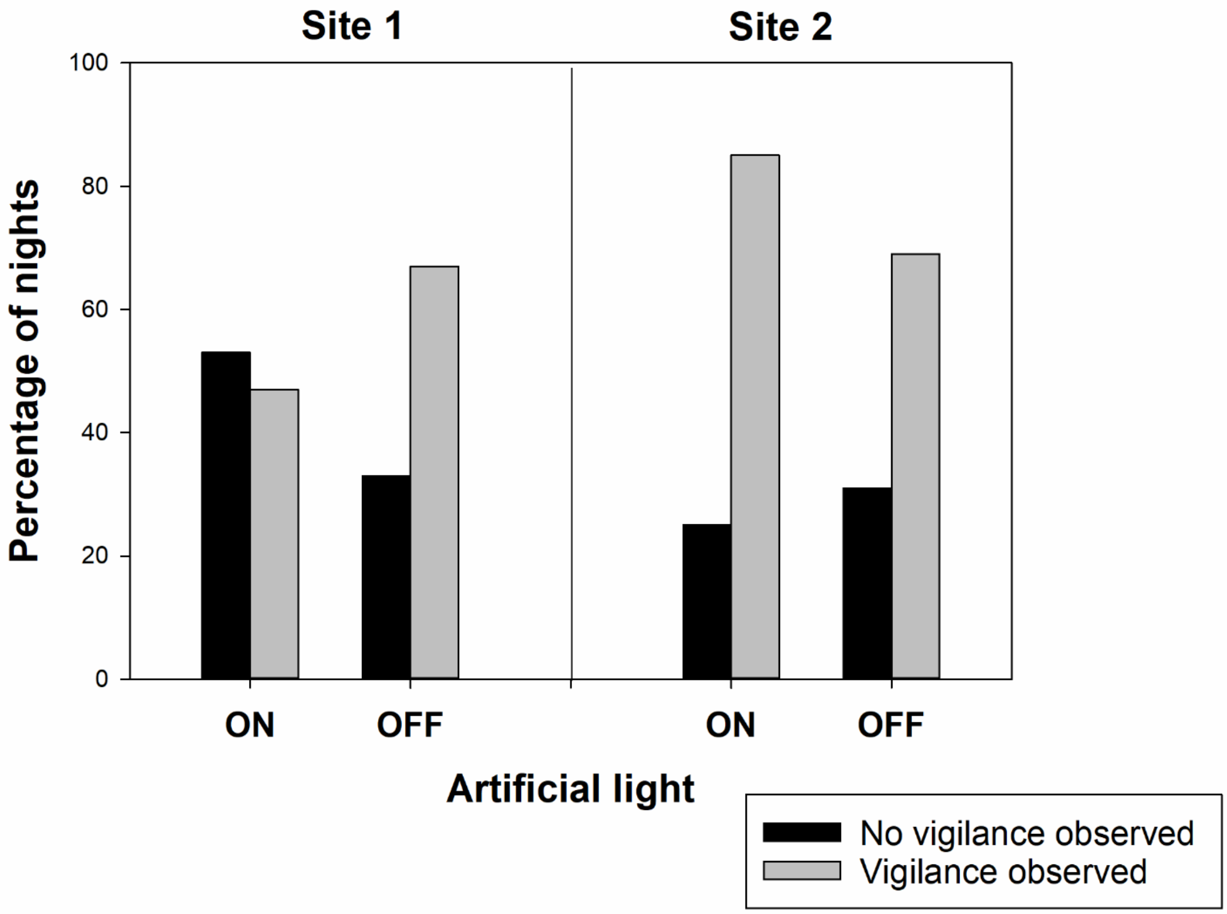

Figure 10 The occurrence of breeding behaviours in relation to the number of little penguins present at the time of occurrence.

CHAPTER 5 - CONCLUSION

Artificial light and anthropogenic noise can have overlapping impacts as these pollutants commonly occur together. As seabirds are highly dependent on their senses to perform tasks (Rajpar 2018), exposure to both introduced light and noise can intensify the stress response of an individual (Rodríguez et al. 2015; Syposz et al. 2021). Wilson et al. (2021) conducted a study investigating the interactive impact artificial light and noise pollution had on avian communities. They found that species that occupied closed habitats were less tolerant of both noise and light exposure than species that occupy mixed and open environments (Wilson et al. 2021). There are no studies that have conducted a study the investigated impacts of both light and noise on seabird species. As seabirds are important bio-indicators that provide a reliable method for researchers to monitor pollutant levels (Elliott et al. 1989; Vander Pol et al. 2009; Piña-Ortiz et al. 2016; Provencher et al. 2010; Rodríguez et al. 2017b; Baak et al. 2020) we are able to use them to monitor the impacts caused by introduced sensory stimulus in the marine environment (Provencher et al. 2010; Dureja & Rathore 2012; Piña-Ortiz et al. 2016; Wilcox et al. 2015; Baak et al. 2020). Changes in their behaviours can lead to increased use of energy reverses (Larcombe 2016), deviate attention (Chan et al. 2010), reduce fitness and breeding success (Ellison & Cleary 1978). While increased pressures on physiological processes can lead to health complications, suppression of the immune system, and altered function of the endocrine system (Barnett & Hemsworth 1990). Therefore, monitoring the behavioural and physiological states of seabirds can indicate the presence of adverse environmental impacts.

Prior to the construction of a new port facility, we wanted to first determine how breeding seabirds on Lipson Island may react to the introduction of artificial light and anthropogenic noise, in a largely agricultural area with low levels of such pollutants. By determining the short-term response of individuals to these stimuli we hoped to get an indication on whether the seabirds would have a negative response to the construction and operations of the port. We achieved this by investigating the behavioural and physiological response of little penguins (*Eudyptula minor*) to construction noise, while assessing the breeding success of little penguins and closely associated species on Lipson Island, the back-faced cormorants (*Phalacrocorax fuscescens*) and the crested terns (*Thalasseus bergii*). We also gathered data on the behavioural response of returning little penguins to artificial light at their landing sites. Our results from the noise experiment revealed little penguins significantly increased their vigilance and took longer to stop exhibiting such vigilance when exposed to construction noises. This showed the behavioural time budget of individuals being impacted leading to reduced time spent resting or preening (Laughlin 2001; Chan & Blumstein 2011; Fanning et al. 2020). We also found that the playback experiment had no negative impact on the breeding success of little penguins, although a longer study that encompassed at least two breeding seasons may have been more appropriate (Australia 2020), with nests exposed to the playback being more successful than those that were not. Our results suggest that exposure to a construction noise can cause individuals to become more alert but did not perceive the short-term presence of the stimulus as a substantial threat. While the results from our light experiment revealed that little penguins arrived later to their landing site on nights when the artificial light had been turned on. Also, little penguins arriving in different sites displayed different levels of vigilance which may be a result of site-specific factors or due to individual personality (Ellenberg et al 2006). We also found, that breeding behaviour occurrence was significantly correlated to conspecific presence. These results indicate that little penguins on Lipson Island were deterred from coming to shore in the presence of artificial light.

Therefore, presence of Port Spencer may lead to long-term exposure of introduced noise and artificial light during seabird breeding season which may cause parents to abandon their nests (Viblanc et al. 2016). Noise produced from the construction and operational phase of the port could increase vigilance behaviours of seabirds incubating or guarding their chicks on land (Franks 2017; Buxton et al. 2017a). This could lead to reduce energy reserves (Larcombe 2016) during a time they should be resting or taking care of their off-spring. While parents that are out fishing during the day may be deterred from returning to the island at night under artificially illuminated conditions. If the foraging partner does not come back to relieve the partner at the nest, it may lead to nest abandon where eggs or chicks may be lost. To prevent such a scenario from occurring at Lipson Island, continued monitoring of seabird behaviours both during the day and at night while the port in being constructed and transitions into its operational phase. This is so researchers can monitor if and when individuals start to become significantly impacted by the port construction or operation.

Management of the Lipson Island seabird colonies should involve an immediate implementation of a monitoring program to observe colony numbers (Schreiber 2000). Monitoring this transition will not only help with the conservation of the seabirds on Lipson Island but can provide valuable information that can be used in future environmental impact studies of coastal developments. As this kind of research is lacking for seabirds. Fortunately, the Lipson Island seabird colonies are not exposed to terrestrial predators but restriction of human activity on the island, especially during the summer months is needed to prevent direct disturbance to seabird nests (Schreiber 2000). This would mean appointing a dedicated ranger for this region where there currently is none. For any management plan to work smoothly all stakeholders must be aware of the impacts artificial light and anthropogenic noise can have on breeding seabirds and work together to reduce the impacts (Yorio 2009). The results from this thesis should be used to help inform guidelines of future management programs.

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