CONSOLIDATED PAPER

Consistent offshore artificial light at night near the last breeding colony of a critically endangered seabird

Johannes H. Fischer1,2 | Igor Debski2 | Graeme A. Taylor2 | Heiko U. Wittmer1

1School of Biological Sciences, Victoria University of Wellington, Wellington, New Zealand
2Aquatic Unit, Department of Conservation, Wellington, New Zealand

Correspondence
Johannes H. Fischer, School of Biological Sciences, Victoria University of Wellington, PO Box 600, Wellington 6140, New Zealand.
Email: johannesfischer@live.nl

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Abstract
Artificial light at night (ALAN) is considered a major threat to biodiversity, yet impacts of offshore ALAN on seabirds remain poorly understood. Particularly understudied are deck strikes (collisions of seabirds with vessels due to ALAN-induced disorientation). To infer deck strike risks to the critically endangered Whenua Hou Diving-petrel (Pelecanoides whenuahouensis; WHDP), we estimated nightly vessel counts and associated probability of floodlight use in close proximity of their only breeding colony, number of WHDP commutes, duration of phenophases, and ultimately the cumulative exposure to floodlights using a Bayesian framework. The estimated nightly number of vessels was 0.76 (0.64–0.89), 31% (21%–38%) of which used floodlights. WHDPs were potentially exposed to 28.9 (19.7–39.7) floodlights per breeding period. There was no evidence for inter-annual variation in vessel counts, floodlight use, or WHDP activity, despite varying environmental conditions. WHDP activity, and therefore risk, was highest during courtship and post-guard. Yet, there was potential for deck strikes throughout the breeding period, which could impact WHDPs. We identify key research questions (likelihood of a deck strike occurring, survival of deck-struck birds, and post-release survival of deck-struck birds). In the meantime, small behavioral changes of vessel operators could be a successful approach to protecting seabirds from offshore ALAN.

Tuhinga whakarāpopoto

Ko te rama tāwhai ite pō (ALAN) he mea whakamōrea ake i te rerenga rauropi, engari ko ngā panga ote ALAN ki ngā manu moana ki tai kāore anō kia aha whakamāramatia. Ko ngāāhuatanga kāore anō kia aha tātari ko te tukinga kāraho (ko te tūtukinga o ngāmanu moana e tāhurihuri ana ki ngā waka, nā te ALAN). Ki te whakapae atu i temōrereatanga o te tukinga kāraho ki te Kuaka Whenua Hou (Pelecanoides whenuahouensis; WHDP) he manu e ngaro haere ana tōna whare ora, i whakatau tatate tini o ngā waka e tere i te pō me te tūponotanga e hāngai ana ki te twhakamahinga o te tūrama raharaha e pātata ana ki pürei kōhanga anake ō rātou,te maha o ngā rerenga o ngā WHDP, te roa o te tauwhi māta, ā, i te otinga kuawhakamahia te anga...
Artificial light at night (ALAN) is a global threat that affects a vast range of taxa including microbial communities, plants, invertebrates, and vertebrates in a variety of ways (Longcore & Rich, 2004; Sanders, Frago, Kehoe, Patterson, & Gaston, 2021). Impacts of ALAN are pervasive and highly diverse, ranging from effects on organismal physiology (e.g., changes to hormonal levels) and activity patterns (e.g., reduction or cessation of certain behaviors) to effects on life history traits (e.g., decreases in orientational ability) and populations (e.g., decreases in abundance) (Sanders et al., 2021). Given the severity, geographic range, and phylogenetic breadth of the impacts of ALAN, addressing this threat should be considered a primary research focus in the 21st century (Davies & Smyth, 2018). Unlike global change phenomena such as increasing CO₂ levels and rising temperatures, the increase in ALAN is unprecedented in the natural world (Gaston, Duffy, Gaston, Bennett, & Davies, 2014). While impacts from ALAN on biodiversity are clearly of global concern, research into the effects of ALAN remains challenging, particularly at the population level (Gaston, Visser, & Höller, 2015).

Seabirds are one of the most threatened species groups on the planet and their populations are affected by numerous threats, including ALAN (Dias et al., 2019). The effects of onshore ALAN on seabirds have received considerable attention (Rodriguez et al., 2017; Rodriguez, Rodriguez, & Negro, 2015). The biggest effect of onshore ALAN on seabirds is disorientation, which can result in birds landing on land instead of water. Grounded seabirds are then vulnerable to collision with vehicles, predation by (non-native) predators, and dehydration (Rodriguez et al., 2012). Globally, rescue centers mitigate the effects of seabirds grounded by terrestrial ALAN (Rodriguez et al., 2017). However, offshore ALAN (e.g., floodlights on vessels or oil platforms) also poses threats to seabirds, particularly in the vicinity of seabird colonies during the breeding period (Black, 2005; Merkel & Johansen, 2011; Ronconi, Allard, & Taylor, 2015). Offshore ALAN can disorient birds and lead to collisions with structures and vessels (i.e., deck strikes). Smaller Procellariiform seabirds, such as diving-petrels, are particularly at risk from deck strikes and several hundred individuals can succumb to deck strikes in a single night (Black, 2005; Holmes, 2017; Ryan, 1991). Deck-struck individuals can die immediately upon hard impacts, or later, due to injuries, contamination with onboard chemicals, or hypothermia following the loss of waterproofing (Black, 2005). Despite clear negative effects, and in contrast to onshore ALAN, risks from offshore ALAN are rarely quantified (Rodriguez et al., 2017). Challenges associated with assessing impacts of ALAN on populations (e.g., confounding factors, the need for long-term studies, and limited knowledge on the origin of individuals; Longcore & Rich, 2004; Gaston et al., 2015, Syposz, Gonçalves, Carty, Hoppitt, & Manco, 2018) are further exacerbated at sea by additional logistical and resource constraints. Consequently, few studies investigate intra- or inter-annual variation in offshore ALAN (Merkel & Johansen, 2011). Understanding intra-annual variation...
of offshore ALAN is crucial, as the vulnerability of species may vary in relation to their breeding phenology and age. For example, adult seabirds may be more vulnerable to offshore ALAN during the courtship or chick-rearing stage when seabirds visit their breeding colonies more regularly, compared to the incubation stage, when commuting is limited as birds are brooding eggs (Taylor et al., 2020). Fledging juveniles have been shown to be more vulnerable to offshore ALAN than adults (Atchoi et al., 2020; Rodríguez et al., 2014).

Understanding inter-annual patterns in dynamic threats such as offshore ALAN is equally important. As marine resources important to humans and seabirds alike move in relation to annually varying environmental conditions (e.g., El Niño Southern Oscillation cycle; ENSO; Schreiber & Schreiber, 1984), presence of fishing vessels and thus the associated offshore ALAN may also vary. Insights into both intra- and inter-annual variation in exposure of species to offshore ALAN are thus of crucial importance to understanding the extent of this threat and designing adequate countermeasures.

The Whenua Hou Diving-petrel (*Pelecanoides whenuahouensis*; WHDP) is a critically endangered seabird (BirdLife International, 2020). The WHDP was historically widespread throughout southern Aotearoa (New Zealand), but following multiple local extirpations caused by non-native predators, the species now only survives on Whenua Hou (Codfish Island; Taylor, 2000) where these predators have been eradicated since 2000 (McClelland, 2002). The WHDP population is currently estimated at 207 (182–235) adults (Fischer, 2020), all of which breed in a minute (0.018 km²) colony in fragile foredunes (Fischer, Debski, et al., 2018; Fischer, 2020). While some onshore threats have been identified (e.g., storms, storm surges, climate change, and competition with congeners), little is known about offshore threats during the breeding period, including offshore ALAN (Fischer et al., 2020; Fischer, Debski, Spitz, Taylor, & Wittmer, 2021). This is of conservation concern, as diving-petrels are among the most frequently recorded taxa in deck strikes caused by offshore ALAN (Black, 2005; Ryan, 1991), including in the waters surrounding Aotearoa (Abraham & Richard, 2019; Fischer, Debski, et al., 2018;
Holmes, 2017). Specifically, ≥374 deck-struck diving-petrels were recorded between 2002 and 2017 in the Stewart Snares Shelf area (the core foraging range of the WHDP during the breeding season; Abraham & Thompson, 2015a, 2015b, 2015c; Fischer, Debski, et al., 2021; Fischer, Wittmer, et al., 2021). While only 5 (1.3%) of these deck-struck diving-petrels were recorded as dead, post-release mortality is poorly understood, at-sea observer coverage varies considerably among fisheries and years, and at-sea observer efforts are directed at quantifying impacts from accidental bycatch rather than impacts from ALAN (Abraham & Thompson, 2015a, 2015b, 2015c; Holmes, 2017). Diving-petrels are also notoriously hard to identify (Fischer, Debski, et al., 2018) and deck-struck diving-petrels are not recorded to species level. Consequently, the impacts of offshore ALAN on the WHDP remain unknown. Anecdotal evidence, however, indicates that deck strike related mortalities can occur in direct vicinity of the last WHDP breeding colony. After offshore ALAN was recorded <1 km from the breeding colony, a dead WHDP washed up and an autopsy indicated spinal traumas (S. Hunter *in litt.* 2020), potentially due to collision impacts. For species persisting in extremely small populations, such as the WHDP, the loss of even a few individuals can be problematic and all threats should be assessed and mitigated to secure species recovery.

We assessed the potential threat from offshore ALAN to the WHDP, while accounting for within and among year variability in offshore ALAN patterns and WHDP activity. Specifically, we counted vessels and associated light use in close proximity (50–1,000 m) of the WHDP breeding colony, recorded WHDP activity, and monitored WHDP burrows across 3 years (2017–2019) subject to varying ENSO conditions. We then used a Bayesian hierarchical framework to estimate (a) the nightly number of vessels, (b) the nightly probability of vessels using floodlights, (c) the nightly number of WHDP commutes, (d) the duration of phenophases, and (e) the cumulative exposure of WHDPs to floodlights per breeding period. Our approach allows for the development of targeted management strategies, which could reduce potential negative effects of offshore ALAN near seabird colonies where and when they are most vulnerable.

## 2 | MATERIALS AND METHODS

### 2.1 | Study site and species

We conducted our study at the last WHDP breeding colony, which is situated within the dunes of Waikoropupu (Sealers Bay; −46.766°S, 167.645°E) on Whenua Hou, Aotearoa (Figure 1). All WHDP burrows are located within the first row of foredunes (i.e., <20 m from the springtide line) and have direct line of sight to the sea (Fischer et al., 2018). WHDPs attend these burrows every 1–4 days between early September and late January, but commute frequency varies by phenphase. WHDP sensitivity to offshore ALAN thus may also vary throughout the breeding period. Whenua Hou is a Nature Reserve with strict access restrictions, but the surrounding waters are not subjected to such legislation (Middleton, 2007). Waikoropupu is one of the few safe anchorages on the western side of Rakiura (Stewart Island). Therefore, vessels often seek shelter and moor overnight in Waikoropupu, creating a potential source of offshore ALAN.

### 2.2 | Vessel and floodlight counts

To infer the risk of deck strikes in close proximity of the WHDP breeding colony, we recorded vessels moored within Waikoropupu and the associated use of floodlights (i.e., bright outward-facing lights, including both fixed floodlights and moveable spotlights, but not cabin or anchor lights) on 188 nights across 3 years (2017: n = 51, 2018: n = 64, 2019: n = 73; referring to the year in which breeding started). These years were subject to varying ENSO conditions. The oceanic Niño Index was −0.79 (La Niña) in 2017, 0.75 in 2018 (El Niño), and 0.43 (approaching ENSO neutral) in 2019 (NOAA Climate Prediction Center; *Climate Prediction Database* repository; available at https://catalog.data.gov/dataset/climate-prediction-center-epcoceanic-nino-index). We conducted counts of vessels and floodlight use during annual capture-recapture efforts of WHDPs, covering their entire breeding period (Fischer, 2020). Capture-recapture efforts were conducted between 21:00 and 02:00 at night when WHDP generally return to their burrows (Fischer et al., 2017). We did not differentiate between different vessel types, as we could only anecdotally identify vessels at night, nor did we record the length of floodlight use per night, or identify the underlying reasons for floodlight use. We allocated records into relevant phenophases based on the mean dates of laying, hatching, and commencement of the post-guard stage per year, which were identified through daily burrow monitoring using a burrowscope (Fischer, Wittmer, et al., 2021).

### 2.3 | WHDP activity monitoring

To infer WHDP activity (nightly number of WHDP commutes on an individual level), we used saltwater-immersion records from geolocators attached to 22 adult WHDPs across three breeding periods (2017–
2019; summarized in Fischer, Debski, et al., 2021). These geolocators (Migrate Technology, Cambridge) did not provide high enough spatial resolution to map WHDP commutes to the breeding colony (two locations per day with ~145 km error; Merkel et al., 2016). However, these geolocators also recorded saltwater-immersion in 10-min intervals (as a value ranging from 0, dry, to 20, fully immersed). This allowed us to infer the nightly number of WHDP commutes from these saltwater-immersion records by assuming that dry periods >1 hr at night during the breeding period correspond with birds attending burrows (Taylor et al., 2020) and, subsequently, quantifying the percentage of saltwater-immersion during the night of interest, as well as the days before and after. For example, if records showed prolonged saltwater-immersion throughout the day, a prolonged dry period at night, and prolonged saltwater-immersion the next day, the inferred number of commutes for the night of interest was 2 (Data S1). Using this approach, we extracted 2,559 counts of nightly commutes across 3 years. We then allocated the nightly number of commutes into phenophases based on burrowscope monitoring (Fischer, Wittmer, et al., 2021).

2.4 WHDP burrow monitoring

To quantify the timing and durations of phenophases, we monitored 25-30 WHDP burrows between early September and late January across three breeding periods (2017–2019) using a burrowscope (Sextant Technology, Wellington; summarized in Fischer, Wittmer, et al., 2021). Specifically, we identified the timing of breeding phenology events (laying, hatching, commencement of post-guard phase, and fledging) by monitoring burrows daily. We identified the timing of arrival at the breeding colony (i.e., initiation of courtship) based on when burrows opened (burrows close over winter due to the mobile nature of sand dunes; Fischer, Wittmer, et al., 2021). We then used the timing of phenology events to quantify the duration of the relevant phenophases: courtship, incubation, guard stage, and post-guard stage.

2.5 Modeling exposure to offshore ALAN

We used a Bayesian hierarchical modeling framework to estimate the nightly number of vessels, the probability of floodlight use, the nightly number of WHDP commutes, the duration of each phenophase per year, and the cumulative number of floodlights per phenophase and breeding period.

First, we modeled the nightly vessel count using a generalized mixed-effects linear model (GLMM) with a Poisson error structure and a log-link function:

$$\log(n_{vessel,i}) = \alpha_{vessel} + \beta_{vessel,k} \times \text{count}_{i,k} + \epsilon_{vessel,y}$$ (1)

in which \(n_{vessel,i}\) is the number of vessels during count \(i\), \(\alpha_{vessel}\) is the intercept of the vessel count equation, \(\beta_{vessel,k}\) is a fixed effect per phenophase \(k\), \(\text{count}_{i,k}\) indicates during which phenophase count \(i\) was conducted, and \(\epsilon_{vessel,y}\) is an annual random effect. We then estimated the nightly number of vessels per phenophase per year \(n_{vessel,k,y}\) using:

$$\log(n_{vessel,k,y}) = \alpha_{vessel} + \beta_{vessel,k} + \epsilon_{vessel,y}$$ (2)

Second, we modeled the probability of floodlight use per vessel using a GLMM with a binomial error structure and a logit-link function:

$$\text{logit}(p_{light,i}) = \alpha_{light} + \epsilon_{light,y}$$ (3)

$$p_{light,i} \sim \text{Bin}(p_{light,i}, n_{vessel,i})$$ (4)

in which \(p_{light,i}\) is the probability of floodlight use, \(\alpha_{light}\) is the intercept of the floodlight equation, \(\epsilon_{light,y}\) is an annual random effect, and \(n_{light,i}\) is the number of vessels using floodlights during count \(i\). We thus assumed that the probability of floodlight use was not subject to intra-annual variation. We estimated the probability of floodlight use per year \(p_{light,y}\) with:

$$\text{logit}(p_{light,y}) = \alpha_{light} + \epsilon_{light,y}$$ (5)

Third, we modeled the nightly number of commutes per bird using a GLMM with a Poisson error structure and a log-link function:

$$\log(n_{commutes,i}) = \alpha_{comm} + \beta_{comm,k} \times \text{comm}_{i,k} + \epsilon_{comm,y}$$ (6)

in which \(n_{commutes,i}\) is the number of commutes for night \(i\), \(\alpha_{comm}\) is the intercept of the commutes equation, \(\beta_{comm,k}\) is a fixed effect per phenophase \(k\), \(\text{comm}_{i,k}\) indicates to which phenophase night \(i\) is allocated, and \(\epsilon_{comm,y}\) is an annual random effect. We then estimated
the nightly number of commutes per bird per phenophase per year \( n_{\text{commutes},k,y} \) using:

\[
\log(n_{\text{commutes},k,y}) = \alpha_{\text{comm}} + \beta_{\text{comm},k} + \epsilon_{\text{comm},y}
\] (7)

Fourth, we modeled the duration of phenophases with another GLMM with a Poisson error structure and a log-link function:

\[
\log(t_i) = \alpha_{\text{duration},k} \times i_{\text{duration},k} + \epsilon_{\text{duration},y}
\] (8)

in which \( t_i \) is the duration of individual phenophase \( i \) in number of days, \( \alpha_{\text{duration},k} \) is the intercept of the phenophase equation, \( \beta_{\text{duration},k} \) is a fixed effect for incubation, guard, and post-guard stages, \( i_{\text{duration},k} \) indicates if phenophase \( i \) is an incubation, guard, or post-guard stage, and \( \epsilon_{\text{duration},y} \) is an annual random effect. We then estimated the duration of each phenophase per year using:

\[
\log(t_{k,y}) = \alpha_{\text{duration},k} + \beta_{\text{duration},k} + \epsilon_{\text{duration},y}
\] (9)

Fifth, we estimated the cumulative number of vessels per phenophase per year \( n_{\text{total vessel},k,y} \) using:

\[
\log(n_{\text{total vessel},k,y}) = \alpha_{\text{total vessel},k} + \beta_{\text{total vessel},k} + \epsilon_{\text{total vessel},y}
\] (10)

We then sampled the cumulative number of floodlights per phenophase per year \( n_{\text{total light},k,y} \) stochastically using:

\[
\log(n_{\text{total light},k,y}) = \alpha_{\text{total light},k} + \beta_{\text{total light},k} + \epsilon_{\text{total light},y}
\] (11)

Ultimately, we summed the cumulative number of floodlights per phenophase to estimate the total exposure of the WHDP breeding colony to offshore ALAN per breeding period \( n_{\text{total light},y} \).

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**FIGURE 2** Estimated number of vessels per night in close proximity (50–1,000 m) of the last Whenua Hou Diving-petrel breeding colony (a), probability of floodlight use (b), number of nightly commutes per Whenua Hou Diving-petrel (c), and an example of a vessel using floodlights (d). Symbols represent posterior means with 95% credible intervals
We fitted our models in the Bayesian modeling platform OpenBUGS 3.2.3 (Spiegelhalter et al., 2014) and used Markov chain Monte Carlo (MCMC) algorithms to obtain posterior distributions. We used vague priors for all intercepts ($N_{\text{mean}} = 0, \text{precision} = 0.001$), fixed effects ($N_{[0, 1]}$), and standard deviations of random effects ($U_{[0, 1]}$), and standard deviations of random effects ($U_{[0, 10]}$). We ran two MCMC chains for 300,000 iterations after a burn-in period of 150,000 iterations. We assessed convergence of chains with the Gelman-Rubin statistic ($\hat{R} < 1.05$) and report posterior means with 95% credible intervals (CI). OpenBUGS code and data are provided in Data S2.

3 | RESULTS

An estimated 0.76 (CI = 0.64–0.89) vessels per night were moored in close proximity of the WHDP breeding colony (Figure 2a). Anecdotal records of individually identifiable vessels ($n = 22$) indicated that vessels were either Rock Lobster ($J. edwardsii$) potting vessels (77%), small inshore trawlers (18%), or set netters (5%). Compared to the courtship period (1.14; CI = 1.02–1.28), the nightly number of commutes was lower during the incubation (0.56; CI = 0.50–0.64; $\beta_{\text{comm,incubation}} = -0.71; CI = -0.82$ to 0.60) and the guard stage (0.86; CI = 0.71–1.03; $\beta_{\text{comm,guard}} = -0.28; CI = -0.47$ to 0.10), while the number of commutes during the post-guard stage was higher (1.57; CI = 1.41–1.75; $\beta_{\text{comm,post-guard}} = 0.32; CI = 0.20–0.43$). There was no clear evidence for inter-annual variation in the nightly number of commutes ($\epsilon_{\text{comm,2017}} = 0.03; CI = -1.05$ to 0.74, $\epsilon_{\text{comm,2018}} = -0.01; CI = -1.10$ to 0.68, $\epsilon_{\text{comm,2019}} = -0.22; CI = -1.33$ to 0.45).

Durations of phenophases varied, but not among years (Table 1; $\epsilon_{\text{duration,2017}} = -0.04; CI = -0.40$ to 0.19, $\epsilon_{\text{duration,2018}} = -0.04; CI = -0.40$ to 0.19, $\epsilon_{\text{duration,2019}} = 0.03; CI = -0.32$ to 0.27). The cumulative number of floodlights per phenophase was estimated as following:

$$n_{\text{total,light,y}} = n_{\text{light,courtship,y}} + n_{\text{light,incubation,y}} + n_{\text{light,guard,y}} + n_{\text{light,post-guard,y}}$$

(12)

TABLE 1  Estimates of durations of Whenua Hou Diving-petrel phenophases in days. Posterior means with 95% credible intervals

| Phenophase          | 2017     | 2018     | 2019     |
|---------------------|----------|----------|----------|
| Courtship           | 27.3 (25.9–28.8) | 27.2 (25.6–28.9) | 29.3 (27.7–30.9) |
| Incubation          | 47.0 (44.7–49.3) | 46.8 (44.2–49.4) | 50.3 (47.8–52.8) |
| Guard               | 8.1 (7.2–8.9) | 8.0 (7.2–8.9) | 8.6 (7.7–9.6) |
| Post-guard          | 38.4 (36.3–40.6) | 38.3 (35.9–40.7) | 41.1 (38.6–43.8) |
| Total breeding period | 120.7 (116.1–125.4) | 120.3 (114.6–120.3) | 129.3 (123.8–134.9) |

4 | DISCUSSION

Our results highlight consistent presence of vessels and associated offshore ALAN on a near-daily basis in close proximity (50–1,000 m) of the last remaining breeding colony of a critically endangered seabird. Despite large variability in environmental conditions associated with climatic oscillations, neither the number of vessels nor...
floodlight use varied among the 3 years of our study. There was also limited evidence for intra-annual variation in the number of vessels, but vessels were more numerous earlier in the WHDP breeding period (i.e., courtship; September–early October), compared to later in the breeding period. Despite these small intra-annual variations, offshore ALAN appeared largely consistent within Waikoropupü, which could lead to WHDP deck strikes.

Deck strikes, induced by offshore ALAN, during the breeding period could have a detrimental impact on WHDP population dynamics. Due to the concentrated nature of the WHDP breeding colony, virtually every breeding adult commutes on a near-nightly basis past vessels. Our results show that, per breeding period, individual adults commute in and out of Waikoropupü approximately 100 times, meaning that over a given breeding period, tens of thousands of WHDP commutes occur past vessels that are potentially using floodlights. Diving-petrels are highly susceptible to deck strikes and have higher mortality rates compared to other seabirds (Black, 2005; Glass & Ryan, 2013; Holmes, 2017). Therefore, deck strikes could reduce adult WHDP survival. This risk appears only present during the breeding period (September–January), as the WHDP non-breeding distribution at sea does not overlap with anthropogenic activity (Fischer, Debski, et al., 2021). However, if a breeding adult succumbs due to a deck strike, this would also result in nest failure, as both parents incubate the egg and raise the chick (Taylor, 2000). This risk is particularly evident during the chick-rearing period when commutes were most frequent. Consequently, deck strikes also have the potential to reduce WHDP productivity. Additionally, as seabird fledglings are disproportionately affected by onshore ALAN (Rodriguez et al., 2017; Wilhelm et al., 2021), the potential for deck strikes could have implications for WHDP juvenile survival. Consequently, offshore ALAN has the potential to impact every key demographic parameter (Gaston et al., 2015) and as such, may be a hitherto overlooked threat to this critically endangered species.

Further research is urgently required to quantify the impacts of offshore ALAN on seabirds, including the WHDP. Researching the impacts of ALAN is challenging (Gaston et al., 2015), but research on offshore ALAN appears particularly challenging. We combined data from disparate sources to provide initial insights into the impacts of this poorly understood threat on WHDPs. However, many pressing research topics on impacts of offshore ALAN remain, as we were not fully able to link our disparate data. Required research avenues that could extend our current understanding mirror the research needs for many other seabirds worldwide (Rodriguez et al., 2017). Research topics for the WHDP include quantifying (a) the duration of floodlight use per vessel, (b) the temporal overlap between commutes of birds and floodlight use, (c) the likelihood of a deck strike occurring due to offshore ALAN (and how this likelihood varies intra-annually under different light sources and weather conditions), (d) the survival probability of deck struck birds, (e) the post-release survival probability of deck struck birds, and (f) the full extent of marine activity generating offshore ALAN and associated deck strike risks across the larger seascape beyond the direct vicinity of the breeding colony. The first two could be answered onshore by combining time-lapse cameras and microchip readers (Fischer et al., 2017) or by utilizing radar technologies (Day, Rose, Prichard, & Streever, 2015). However, insights into the latter four would require concerted data collection efforts at sea. Combining high-resolution GPS tracking and light-recording geolocators could provide novel research opportunities (Dupuis et al., 2021; Krüger, Paiva, Petry, & Ramos, 2017). Deck strikes receive considerably less attention than other offshore threats, such as accidental bycatch in commercial fisheries, for which regular summaries and risk assessments based on standardized observer data across large spatial and temporal scales are available (Abraham & Richard, 2019). Instead, records of deck strikes are either anecdotal (Black, 2005) or only collected as part of observations made during commercial fishing (Holmes, 2017). Given the current uncertainty surrounding offshore ALAN and the impacts on seabirds, heightened attention towards this threat and improved data collection in the seas surrounding Aotearoa and throughout the world’s oceans is urgently required.

Targeted management of offshore ALAN could have considerable benefits for the WHDP and many other seabirds. Periodic spatial restrictions of vessels within Waikoropupü, or around all of Whenua Hou, could be a potential solution to mitigate impacts from offshore ALAN. Spatial restrictions in nearshore waters to protect important island ecosystems are in place, for example, around Gough and Tristan da Cunha (Requena et al., 2020). Spatial restrictions aimed at mitigating deck strikes could be limited to the WHDP breeding period (September–January), or even to high activity periods only (i.e., during courtship from September to early October and during chick-rearing and fledging from December to January). Spatial restrictions of vessels, however, could impact the values and interests of tangata whenua (people of the land) and fishing operators (e.g., access to safe anchorage and/or fishing grounds). Instead of blanket spatial restrictions, the encouragement of behavioral changes of vessel operators around seabird colonies during the breeding period could be another solution to reduce the impacts of offshore ALAN (Glass & Ryan, 2013). Behavioral changes that should be encouraged include: (a) the reduction of external deck lighting, flood lights in
particular, to an absolute minimum (i.e., for navigational purposes only), (b) the shielding of external deck lights to illuminate only areas of interest (i.e., the deck), (c) the use of black-out blinds to reduce offshore ALAN caused by cabin lights, and (d) the provision of protocols on safe handling, treatment, and release procedures of deck-struck birds to vessel operators (Black, 2005). Such behavioral changes could reduce offshore ALAN to the times when absolutely required (i.e., navigation). These small but targeted behavioral changes could be an easy, yet highly efficient compromise that acknowledges divergent values at sea, while successfully addressing the risks of offshore ALAN to seabirds, especially near their breeding colonies at times when they are most vulnerable.

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CONFLICT OF INTEREST
The authors declare no potential conflict of interest.

AUTHOR CONTRIBUTIONS
Johannes H. Fischer conceptualized the study; Igor Debski, Graeme A. Taylor, and Heiko U. Wittmer supervised the project; Johannes H. Fischer collected the data; Johannes H. Fischer analyzed the data; Johannes H. Fischer wrote the manuscript; Igor Debski, Graeme A. Taylor, and Heiko U. Wittmer edited the manuscript.

ETHICS STATEMENT
Access to Whenua Hou and methods were approved by the Whenua Hou Komiti, Kaitiaki Rōpū, the Victoria University of Wellington Animal Ethics Committee (VUW AEC 22252, VUW AEC 23283, and VUW AEC 27621), and the New Zealand Department of Conservation (M1718/01, M1819/01, and M1920/02).

DATA AVAILABILITY STATEMENT
All data are available in supplemental materials.

ORCID
Johannes H. Fischer https://orcid.org/0000-0003-3527-1671

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**SUPPORTING INFORMATION**

Additional supporting information may be found online in the Supporting Information section at the end of this article.

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