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Range shifts and the population dynamics of tropical, subtropical, and rare fishes in New Zealand

A thesis submitted in partial fulfilment of the requirements for the degree of Doctor of Philosophy in Ecology at Massey University, Auckland, New Zealand

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Thesis abstract

The rate of species re-distribution during the Anthropocene is unprecedented. The expectation is that as global temperatures continue to rise, tropical regions will become increasingly inhospitable, and temperate regions will become more tropicalised and biodiverse as species track favourable conditions poleward. Climate change mediated range shifts are causing the distinctness of species assemblages and biogeographic regions to erode, and the most significant biodiversity changes are currently occurring in the coastal marine environment. Identifying the species currently undergoing range shifts and predicting where and when future climate-mediated range shifts will occur is critical to proactively manage changes in resource-based human livelihoods and meet conservation goals. However, identifying range shifts is often hampered by a lack of baseline distributional data. In the marine environment, large areas remain under-surveyed, and given that marine species are often cryptic, wide-ranging, and highly mobile, our knowledge of geographic distributions is far from complete. This thesis aimed to fill this knowledge gap by using a combination of novel data sources and methods to set an accurate baseline for the spatio-temporal distribution of tropical, subtropical and rare teleost fishes in a temperate marine setting and develop methods to allow us to monitor future biodiversity change. Teleost fishes are valuable indicators of current and future change; they are early responders to climate-mediated ocean warming and are charismatic and highly visible, increasing the ease of monitoring, particularly by citizen scientists. First, data sourced from published accounts, scientific surveys, commercial catches, and citizen science sources were examined to determine the spatio-temporal distribution of tropical, subtropical, and rare fish in NZ waters. I characterise their contribution to New Zealand's marine biodiversity and set a baseline for future monitoring of climate-driven biodiversity changes. I found that the contribution of tropical, subtropical, and rare fishes to New Zealand biodiversity is significant, and their occurrences and diversity have increased over the past 50 years. Second, I present and test a novel method that combines citizen science with expert knowledge to classify out-of-range occurrences for marine fishes as potential range extensions or human-mediated dispersal events. The stepwise approach uses qualitative decision making and scoring tools to classify citizen science observations of tropical, subtropical and rare fishes and combines these classifications with expert validation to increase confidence. By applying the method to a range of focal species, I successfully identified species that had undergone range shift into or within New Zealand waters and one species whose range shift was facilitated by human-mediated dispersal. The ease of our approach and the intuitive outputs should appeal to managers and science practitioners concerned with climate-induced biodiversity changes and alien species detection. Third, I apply the classification methods and distributional baselines from Chapters 2 and 3 to citizen science occurrence data for tropical, subtropical and rare fishes in New Zealand waters to identify those species that are tohu (indicators) of change and identify the areas where biogeographic change is occurring. Labrids and Pomacentrids disproportionately contributed to out-of-range occurrences, and 87% of all out-of-range occurrences represented potential and actual range extensions. Locations with a high occurrence of potentially range shifting fishes were centred in north-eastern New Zealand, with occurrences of potentially range shifting fishes decreasing with increasing latitude. I surmise that climate-mediated biodiversity change in New Zealand fishes is occurring and present a list of species contributing to range shifts by region to facilitate effective monitoring and impact mitigation. Overall, this thesis contributes new knowledge regarding the rate of poleward climate-mediated range shifts of marine teleost fishes to New Zealand and a suite of novel tools to facilitate future monitoring and impact mitigation of climate-mediated range shifts. I successfully demonstrate the capacity of ocean-going citizen scientists to identify range shifts in the marine environment and overcome some of the common biases and sampling errors associated with citizen science data by incorporating expert validation and knowledge. My results suggest that New Zealand is currently not a hotspot for climate-mediated biodiversity shifts but biodiversity shifts are occurring and the contribution of tropical, subtropical and rare fishes to New Zealand diversity is increasing. These contributions of this thesis will provide a baseline and framework for resource managers,

science practitioners and citizens, to monitor range shifts and manage the impacts of climate change on New Zealand's marine ecosystem.

Declaration by Author

The research carried out for my doctoral thesis has been used in whole or in part for this qualification only. The research is my original work, except as indicated by appropriate attribution in the text and/or acknowledgements; quotation marks have been used where required; and I take responsibility for the content and quality of this thesis. I have clearly stated the contribution by others in jointly authored works, which can be found at the end of each chapter. The "Statement of Contribution to Doctoral Thesis Containing Publications (DRC16)', has been completed for each published article within the thesis, and is included in the electronic copy at the end of each chapter.

Publications during candidature

Technical reports

Middleton, I; Pande, A; D'Archino, R & Middleton, C. 2021. Second delimiting survey of the non-native alga, *Caulerpa brachypus*, at Aotea/Great Barrier Island. NIWA Client report for Biosecurity New Zealand.

Pande, A; **Middleton, I** & Middleton, C. 2021. Delimiting survey of the non-native alga, *Caulerpa brachypus*, at Aotea/Great Barrier Island. NIWA Client report. NIWA Client report for Biosecurity New Zealand.

Peer-reviewed journal articles included in this thesis

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Table of contents_Toc100494251

1 General introduction

The rate of environmental change during the Anthropocene has been unprecedented, and the impacts have been documented on every continent, in every ocean, and in most major taxonomic groups (Rosenzweig et al., 2008). Changes in the geographic distribution, or range, of organisms, have been reported globally as a ubiquitous indicator of climate change (Parmesan & Yohe 2003; Poloczanska et al. 2013). The expectation is that as global temperatures increase, tropical regions will become increasingly inhospitable, and temperate regions will become more tropicalised and biodiverse as species track favourable conditions poleward (Molinos et al., 2016).

The rate and magnitude of climate-mediated biodiversity change in the coastal marine environment has far exceeded those encountered in terrestrial environments (Burrows et al., 2011; Poloczanska et al., 2013). Most marine organisms are ectotherms, making them highly sensitive to changes in local climate (Heath et al., 2012; Pinsky, Selden, & Kitchel, 2020; Pörtner & Peck, 2010; Sunday, Bates, & Dulvy, 2012). Furthermore, the rate of isotherm shifts and shifts in seasonal therma maxima have been significantly greater in the ocean than on land (Burrows et al., 2011). Consequently, marine range shifts are occurring approximately four times faster than observed rates on land (Burrows et al., 2011; Chen, Hill, Ohlemüller, Roy, & Thomas, 2011; Poloczanska et al., 2013; Sorte, Williams, & Carlton, 2010). Despite the speed and pervasiveness of climate-mediated range shifts in the marine environment, individual species responses can be variable and difficult to predict even in regions where the rate of environmental change has been relatively consistent (Dulvy et al. 2008, Pinsky et al. 2013).

Identifying where and when climate mediated range shifts will occur is critical if we are to proactively manage changes in resource-based human livelihoods and meet conservation goals. However, confidently detecting range shifts requires a thorough understanding of the geographic range that species occupy (Gledhill et al., 2015). Given that large areas of the marine realm remain under-surveyed (Hortal et al., 2015) and that marine species are often cryptic (Hubert et al., 2012), wide-ranging, and highly mobile (e.g., Block et al., 2005; Bonfil et al., 2005), our knowledge of geographic distributions is far from complete (Appeltans et al., 2012; Hortal et al., 2015). Systematic surveys of a species' range at appropriately broad spatial and temporal scales are costly and time-consuming; hence, marine scientists and managers must often rely on diverse sources of knowledge (Delaney, Sperling, Adams, & Leung, 2008).

Citizen science has been a valuable tool for increasing temporal and spatial data resolution and detecting range shifts in marine settings. Long-term biodiversity monitoring at scales appropriate to detect range shifts is costly, time-consuming, and often spatially restricted (Hortal et al., 2015; Middleton et al., 2021). Citizen science- the participation by citizens, who are not usually trained scientists, in scientific projects-has been successfully used to overcome a lack of baseline data, increase the monitoring capacity of science programmes and detect out-of-range occurrences in marine settings (Pecl et al., 2019; e.g., Range Extension Database and Mapping Project [Redmap] Reef Environmental Education Foundation [REEF]; European Alien Species Information Network [EASINCitizen science programmes]). Even though citizen science data has proven utility in contributing toward scientific (e.g., Lenanton et al., 2017; Soroye, Ahmed, & Kerr, 2018) and policy outcomes (e.g., Delaney et al., 2008; Madin et al., 2012), the observational capacity of NZ's ocean going public has not yet been exploited.

This thesis contributes novel approaches, tools, and empirical data to the growing fields of marine biogeography and citizen science and expands our understanding of the impacts of climate change on marine diversity. The goals of this thesis were to develop methods that allow for the incorporation of novel data sources to augment scientific knowledge of species ranges and identify range shifting species in a temperate oceanic island setting. In particular, this thesis focuses on the benthic and pelagic teleost fish fauna of New Zealand. I present an introduction to species ranges and the factors limiting species distributions, the impact of climate change on species ranges in terrestrial and marine ecosystems, and the utility of citizen science in monitoring range shifts

(Chapter One). Chapters Two, Three and Four present the current state of knowledge regarding tropical and subtropical fishes in New Zealand waters, the development of a methodology that allows for the integration of opportunistic sightings into analyses of marine range shifts, and the application of this method to identify novel range shifts in New Zealand, respectively. A more detailed outline of the chapter contributions to the overall goals of this thesis is included at the end of this chapter.

1.1 Geographic distributions and species ranges

Species ranges are a fundamental unit in ecology and attempting to define what constitutes a species range has been a long-standing area of interest (e.g., Darwin 1859, MacArthur, 1972 & Böhm et al., 2017). At its most fundamental definition, a species range is the geographic area where a species occurs during the entirety of its lifetime, including the areas where individuals or groups migrate. Species ranges vary by about 12 orders of magnitude, from a single freshwater spring (i.e., the Devils hole pupfish) to a range encompassing the entirety of the world's oceans (i.e., the Blue Whale). Within a taxonomic grouping, closely related species often have similar-sized ranges (Gaston, 1991) and most species will have small to moderate-sized ranges, with large ranges being an exception (Gaston, 1991, 1996; Rapoport, 1982; Willis, 1922). In 1975 Eduardo Rapoport identified several consistent patterns in range sizes of terrestrial species. For example, range sizes vary consistently across ecogeographic gradients, with range sizes increasing with latitude (Stevens, 1989) and elevation (Stevens, 1992). Stevens (1989), who coined the pattern "Rapoport's rule", found that it holds true at higher taxonomic groupings. Subsequent examination of Rapoport's rule in marine environments has found that range sizes also increase consistently along depth gradients (Macpherson & Duarte, 1994; Stevens, 1996). Most studies that have tested Rapoport's rule have supported his hypothesis, and species with the smallest range sizes are consistently found in tropical areas, at low elevations and shallow depths (Gaston, 1996).

Understanding the factors, and the interactions between factors, setting range boundaries is crucial in predicting ecosystem responses to anthropogenic change. Although geographic features such as rivers, oceans or mountain ranges can set range limits, species are rarely spread across the entirety of the space available to them (Sexton, McIntyre, Angert, & Rice, 2009). Essentially species' ranges are limited when the rate of emigration and mortality exceeds immigration and birth (Gaston, 1990; Holt, Keitt, Lewis, Maurer, & Taper, 2005). Therefore, the geographic area a species occupies is a spatial expression of a species' ecological niche and environmental tolerances and is defined by a range of abiotic and biotic factors (Lomolino, Riddle, & Brown, 2006; Sexton et al., 2009). Bartholomew (1958) was one of the first to hypothesise that a species distribution was limited by the environmental factor for which a species had the narrowest tolerance. It is now generally accepted that species with broader fundamental niches will attain higher densities locally, tolerate wider ranges of environmental extremes, and occupy larger geographic areas than species with narrow, more specialised niches (Gaston & Spicer, 2001). Although this pattern underlies much of the contemporary hypotheses in biogeographic research, it may be overly simplistic (Sexton et al., 2009). Increasingly, empirical evidence is demonstrating that range sizes and the location of range boundaries are determined by complex interactions between a species dispersal ability, their ecological niche (Helmuth, Kingsolver, & Carrington, 2004), biotic interactions, habitat availability and environmental variability (Eckhart et al., 2011; Lynch et al., 2014).

A long-standing theory, originating with Darwin, suggests that abiotic forces set species range limits at high-stress areas like high latitude, high elevation, and temperature extremes, while species interactions set range limits in apparently more temperate regions. Competition for resources and space is one of the best described biotic interactions that limit species ranges (Case, Holt, McPeek, & Keitt, 2005; Sexton et al., 2009). Most of the existing evidence of biotic interactions setting geographic range limits relies on correlations between the abundances of competitors or predators and prey. However, simply observing negative associations does not exclude that species simultaneously respond to underlying abiotic variability or gradients (Sexton et al., 2009). The most definitive empirical evidence that biotic interactions play a role in setting range limits hails from experimental manipulations where abiotic factors can be controlled. For example, Wethy (2002) transplanted a barnacle species (*Chthamalus fragilis*) beyond its northern range limit and demonstrated it could only survive in the absence of a known competitor (S*emibalanus balanoides*). Similarly, the experimental exclusion of grazers at lower depth limits allowed a turf forming algae (*Mazzaella parksii*) to extend its depth range (Harley, 2003). The rapid population growth of introduced species transported outside of their historical range by anthropogenic vectors, and away from their usual competitors, is another example that biotic interactions limit species ranges (Case et al., 2005).

All biotic interactions that impact range limits take place within a landscape of varying abiotic conditions. For any given species, population growth will become negative at some point along an environmental gradient (Gaston, 1990; Sexton et al., 2009). This point will generally correspond with a mismatch between environment and fundamental niche limits and is recognised as the limit of a species range (Sexton et al., 2009). Traditionally species distribution or abundance have been used as a proxy for how abiotic or environmental factors set species range limits. However, correlations between distribution or abundance and environmental drivers can result merely by coincidence and do not inform the mechanisms underlying the correlations (Higgins et al. 2012a). Species respond in both a gradual and a threshold manner to environmental variables (Sexton et al., 2009). Gradual responses affect species physiology, growth, phenology, and reproductive output of a species, whereas threshold responses are the responses to extremes; in these scenarios, broad-scale mortality, population declines, or dispersal will occur (Sexton et al., 2009). Therefore, abundance itself is not always the most accurate measure of abiotic controls of species distributions. For example, the high abundance of a population composed only of juveniles does not constitute a self-sustaining population, and the high abundance might obscure the fact that the environment limits a species distribution or range. Combining abundance measures with the variation in physiological stress and individual traits is a more appropriate approach to estimating the impact of abiotic factors on species distributions. For example, the concentration of thermal stress proteins along a temperature gradient can demonstrate the relative importance of temperature in setting range limits for that species (Feder & Hofmann, 1999). However, recently concurrent range shifts of multiple, unrelated species in response to climate change and global warming suggest that ambient temperature may be a ubiquitous factor setting range limits, particularly in the marine environment (Parmesan & Yohe, 2003).

1.2 Climate change and species ranges

The rapid and pervasive impacts of anthropogenic climate change have renewed interest in the role of ambient temperature in determining species distributions and global biodiversity. In the last 100 years, climate change has resulted in an average global temperature increase of 0.6°C, and an average temperature anomaly that was the warmest on record, +0.82°C above the average for the 20th century (NOAA, 2021). Additionally, climate change has resulted in broadscale isotherm shifts and increased climate anomalies (NOAA, 2021). The expectation is that species will track optimal conditions or 'climate envelopes', poleward or to increased altitudes and depths, as tropical and low altitude regions warm beyond thermal tolerances (Parmesan et al., 1999; Parmesan & Yohe, 2003; Poloczanska et al., 2013; Walther et al., 2002). This expectation has been supported by evidence of poleward and upward (or deeper) range shifts for a range of taxonomic groups and geographic settings (Hill et al., 2002; Parmesan et al., 1999; Walther et al., 2002), with 74-91% of species that have shifted moving in a direction that corresponds with shifts in isotherms and climate change observations (Parmesan & Yohe, 2003).

Although the impact of climate change on species redistribution at global scales has been posited (Beaugrand, Edwards, Raybaud, Goberville, & Kirby, 2015; Cheung, Watson, & Pauly, 2013; Parmesan & Yohe, 2003), the rate of environmental change and subsequent range shifts are spatially heterogenous (Burrows et al., 2014; Chen, Hill, Ohlemüller, Roy, & Thomas, 2011; Parmesan et al., 1999; Sorte, Williams, & Carlton, 2010). For example, global temperature trends indicate broad-scale warming in tropical areas during the last two decades, whereas warming trends at high latitudes have been more variable (NOAA, 2021). At more localised scales, observed and projected changes in environmental variables such as temperate and precipitation also demonstrate significant spatial variation (Dunn et al., 2020; Feng et al., 2014; IPCC, 2013). For example, globally several regions have experienced, or are predicted to experience, higher than average variability or increases in climate variables including temperature (Diffenbaugh & Giorgi, 2012). These climate change hotspots often correspond with areas that have the highest biodiversity globally, including equatorial Africa, Southeast Asia, and the Amazon basin (Fan, Miao, Duan, Shen, & Wu, 2021; Turco, Palazzi, von Hardenberg, & Provenzale, 2015).

Species in tropical climate change hotspots are likely to be disproportionately vulnerable to climate-mediated population declines or range shifts (Wiens, 2016). Unlike species at high latitudes that experience unstable conditions as a norm, the climate in tropical areas has been relatively stable, and species are often highly specialised and have small ranges (Stevens, 1989). For similar reasons, highly specialised and endemic species are particularly prone to climate change as even small shifts in temperature or environmental variables can result in physiological stress, reduced reproductive output and rapid population declines (Bellard, Bertelsmeier, Leadley, Thuiller, & Courchamp, 2012). If climate change mediated losses of specialist and tropical species continues, the distinctness of species assemblages and biogeographic regions may be eroded.

Species have continually adapted to changing climatic variables, however the rate of change during the Anthropocene has been unprecedented (Parmesan, 2006; Parmesan & Yohe, 2003). Species encountering adverse environmental conditions, like climate-induced warming, have three options; they can either adapt in situ, shift along an environmental gradient to more favourable conditions, or undergo local extinctions (Maggini et al., 2011). Current research suggests that poleward range shifts will be the predominant response to climate change (Parmesan & Yohe, 2003). Studies have reported increases in tropical plant (Root et al., 2003), butterfly (Parmesan et al., 1999), fish (Hiddink & ter Hofstede, 2008) and bird (H-Acevedo & Currie, 2003; Van Der Wal et al., 2013) diversity in high latitude regions linked to climate-driven poleward range shifts (Hickling, Roy, Hill, Fox, & Thomas, 2006). However, species will respond idiosyncratically to environmental factors including climate mediated warming. Individual species traits and biotic interactions will impact the rate, extent and direction of climate-induced range shifts (Johnson et al., 2011). For example, highly specialised or endemic species with small ranges may not have the capacity to undergo range shifts, and climate change may result in a higher proportion of climate-mediated local species extinctions (Thomas et al. 2004). A recent analysis of the impact of climate change on biodiversity hotspots suggests that extinction of endemic species could reach 43% in the worst-case scenario, representing a potential loss of up to 60 000 endemic species (Malcolm, Liu, Neilson, Hansen, & Hannah, 2006). Climate change mediated range shifts and species losses will also result in novel species interactions, modified food webs and potentially whole regime shifts. The ubiquity of climate-driven range shifts and localised extinctions highlight the need to identify and monitor species and critical locations of change (Hobday, Cochrane, et al., 2016; Pecl, Hobday, et al., 2014), and current research suggests that range shifts are occurring at approximately an order of magnitude faster in the marine environment than in terrestrial systems (Chen, Hill, Ohlemüller, Roy, & Thomas, 2011; Poloczanska et al., 2013; Sorte, Williams, & Carlton, 2010).

1.3 Climate change in marine systems

Climate change has resulted in changes in the physical properties of the ocean that in turn have had implications for marine species, and the ecosystems and communities that rely on them (Hoegh-Guldberg et al., 2014; Poloczanska et al., 2013; Poloczanska et al., 2016; Pörtner & Peck, 2010). The ocean has absorbed over 90% of the extra energy arising from anthropogenic greenhouse gas emissions, which has resulted in sea surface temperature increases of around ~1°C since the beginning of the 20th century (0.89°C, over the period 1901–2012; IPCC, 2013). However, warming has been variable at regional scales; the Indian Ocean (0.11°C per decade) has warmed faster than the Atlantic (0.07°C per decade) and the Pacific Ocean (0.05°C per decade, Hoegh-Guldberg et al., 2014), but some of the fastest-warming rates (between 1950-2009) were observed in the Baltic sea (0.99°C) and the South China Sea [0.80°C, Hoegh-Guldberg et al. (2014)]. Despite regional variation, most oceans have undergone long-term warming or an increase in the warmest or coolest months of the year, with winter and spring temperatures warming fastest in most regions (Hoegh-Guldberg et al., 2014). Regional current systems, isotherms positions, wind patterns, nutrient availability, and salinity, have also changed in response to anthropogenic climate change (Hoegh-Guldberg et al., 2014). Although the rate of warming in the ocean is up to three times lower than air temperatures in terrestrial systems, the rate of isotherm shifts between 1960 and 2009 have been comparable or faster than on land (Parmesan & Yohe, 2003). For example, in the North Sea, a 1°C increase in ocean temperatures over 40 years has resulted in a 217km per decade shift in the 10°C isotherm, and in South Africa, the 27°C isotherm shifted 178km over 19 years (Beaugrand, Luczak, & Edwards, 2009; Lloyd, Plagányi, Weeks, Magno-Canto, & Plagányi, 2012). Under moderate (RCP 4.5) and high (RCP 8.5) emission trajectories, the ocean will continue to warm at similar rates to those described here, potentially resulting in marine species losses, biodiversity shifts, and the loss of ecosystem functions (Poloczanska et al., 2013; Poloczanska et al., 2016).

Despite variable rates of climate-mediated ocean warming, some of the most significant recent biodiversity changes have occurred in coastal marine environments (Burrows et al., 2011; Poloczanska et al., 2013). Most marine organisms are ectothermic, i.e., they lack the ability for internal thermoregulation. Consequently, their physiological functions are directly impacted by changes in ambient temperature, making them highly sensitive to climate-mediated warming. Furthermore, marine species generally have high fecundity, high mobility and are often dispersed by ocean currents, increasing their ability to avoid suboptimal environmental changes by dispersing to new, more suitable areas (Pinsky et al., 2020; Pinsky, Worm, Fogarty, Sarmiento, & Levin, 2013). The general expectation is that the responses of marine species to climate change will include poleward range shifts, changes in the timing of major phenological events such as reproduction and migration, and a reduction in body size for many marine species (Poloczanska et al., 2013). Recent metaanalyses of climate change impacts on marine systems have provided strong evidence to support these expectations. Marine organisms have, on average, shifted the leading edges of their distributions by 72.0 ± 13.5 km per decade (generally poleward), which is significantly faster than the rate of range shifts in terrestrial systems even at latitudes where historically, the velocities of isotherm migration in the ocean and on land have been similar (Burrows et al., 2011). Poloczanska et al. (2016) also noted that the abundance of 52% of warmer water species increased in response to warming oceans, and 52% of cold-water species declined. These abundance changes linked to ocean warming hold true for varying taxonomic groups across trophic levels. For example, 1400 years of foraminifera samples in the North Atlantic Ocean revealed a significant increase in the abundance of tropical and subtropical species in the last 100 years, which corresponded to the areas undergoing the fastest rate of warming (Beaugrand, Luczak, & Edwards, 2009; Hoegh-Guldberg et al., 2014). Likewise, a cooler water barnacle (*Semibalanus balanoides*) has declined in abundance in the UK and France due to lower recruitment success linked to climate warming (Hawkins et al., 2009).

Climate change is causing the seasons in the ocean to change, winters are becoming warmer, and summer maxima are increasing (IPCC, 2019). Phenological events, including reproduction and migration timing, are linked to seasonality, and evidence suggests that spring phenologies in marine systems have advanced by 4.4 ± 1.1 days a decade (Poloczanska et al., 2013). Some of the most significant advancements were observed in the timing of zooplankton blooms. Fish eggs, fish larvae and echinoderm larvae blooms advanced at rates between 9.5 days and 12.9 days per decade (Edwards & Richardson, 2004), which is significantly higher than the average rate of spring advancement of phenologies in terrestrial environments (2.3 days per decade; Parmesan & Yohe, 2003). Although phenological shifts can impact reproductive capacity and survival, data relevant to this phenomena in the marine environment are sparse and hard to collate retrospectively.

The ecological responses to climate change in the marine environment are varied and many; however, by far, the most reported response is a change in a species distribution (Poloczanska et al., 2016). Climate mediated range shifts can impact commercially important fish stocks, ecologically important conservation areas and the development of policy frameworks [e.g., Dulvy et al. (2008); Ramírez, Afán, Davis, and Chiaradia (2017)]. These broad-scale potential impacts have driven a surge in the research interest surrounding the drivers and patterns of climate-mediated range shifts in the marine environment. Generally, climate mediated range shifts are measured at the leading (poleward or deeper) and trailing (equatorial or shallower) edges of a species range (Bates et al., 2014). The expectation is that climate-mediated warming will lead to range contractions at trailing edges and extensions at leading edges (Bates et al., 2014). At the poleward (leading) edges of distributions, range shifts result from dispersal to a new area and increased ambient temperatures in areas that previously had unfavourable climatic conditions (Bates et al., 2014). However, poleward or leading-edge shifts are variable in both the rate and direction (Pinsky et al., 2013; Poloczanska et al., 2013). The rate and frequency of range shifts are constrained by species-specific traits (e.g., dispersal potential), biotic interactions, habitat constraints in recipient communities, the presence of biogeographical barriers, and differences in regional warming rates (Burrows et al., 2014; Sunday et al., 2015; Poloczanska et al., 2016). Despite the heterogeneous responses of marine species to climate change, evidence suggests that the leading edges of marine species were extending at an order of magnitude faster than in terrestrial systems (Poloczanska et al., 2013). The fastest leading-edge expansions are in highly mobile and pelagic species, including phytoplankton (470km per decade), teleost fishes (277km per decade) and zooplankton (142km per decade; Poloczanska et al., 2013).

Dispersal of pelagic species and the early life stages of benthic species by ocean currents has been implicated as one of the mechanisms facilitating range shifts of plankton, fishes, and invertebrates in both the Atlantic and Pacific Oceans (Beaugrand, Luczak, & Edwards, 2009; Johnson et al., 2011a; Johnson, Ling, Ross, Shepherd, & Miller, 2005; Last et al., 2011; Nye, Link, Hare, & Overholtz, 2009). Marine species often have complex life cycles, and many will spend at least a portion of their life in pelagic environments. For example, many benthic fishes and invertebrates are relatively sessile with small home ranges but have a dispersive planktonic stage (e.g., Leis, 2006; Shanks, 2009). Ocean currents can rapidly advect these pelagic species and larvae away from source populations and facilitate long-distance dispersal to novel recipient communities outside of known range limits. Although long-term variability makes generalisation and causation difficult, evidence suggests that climate change has altered the position and intensity of significant ocean currents systems, with subtropical gyres strengthening and expanding in all the major ocean basins (Hoegh-Guldberg et al., 2014). For example, climate change has been implicated in the strengthening and poleward penetration of the East Australia Current (EAC). The EAC carries warm water and the larvae and juveniles of many tropical species poleward along the coast of Australia. Strengthening of the EAC has resulted in tropical fishes and invertebrates being advected past the leading, or southern, range limits to new areas like Tasmania (Booth, Figueira, Gregson, Brown, & Beretta, 2007; Craig R. Johnson et al., 2011b; Sunday et al., 2015).

Not all out-of-range individuals that arrive in novel recipient communities will persist long term, and even less will have the ability to breed in novel locations (Bates et al., 2014). The ability of a species to disperse to, persist and breed in, novel locations are differentially influenced by species tolerances, environmental variables, and climate change. Pelagic larval duration, propagule pressure, mobility and climate-induced change in the strength and directionality of ocean currents will determine the rate of new species immigration or arrival events (Booth, Figueira, Gregson, Brown, & Beretta, 2007). However, changes in the local marine environment of recipient locations (e.g., sea surface temperatures, productivity) and altered seasonality resulting from climate change can impact the persistence and breeding of new species or previously rare species (Bates et al., 2014; Monaco et al., 2020). The persistence of range shifting species in novel recipient communities is further limited by species-specific traits and resource requirements (Munday et al. 2008a; Cheung et al. 2010). Generalist species have broad physiological tolerances, high phenotypic plasticity, and broad dietary or habitat tolerances, which are expected pre-requisites for persistence, population increase and reproduction in novel recipient communities (Angert et al., 2011; Keith et al., 2011;

Weir and Salice, 2011; Bates et al., 2013; Knutsen et al., 2013). Large geographic ranges characterise generalist species due to their broad environmental, habitat, and dietary preferences (Gilchrist, 1995; Rivadeneira & Fernández, 2005; Bates et al., 2014; Sunday et al., 2015). Globally, the success and competitive dominance of generalist marine species undergoing poleward range shifts has led to a disproportionate decline in specialist marine species diversity (Clavel, Julliard, & Devictor, 2011). This broad-scale homogenisation of biodiversity is particularly evident in temperate regions and areas undergoing rapid warming due to climate change (Richmond et al., 2005).

Globally the impacts of climate change, and the subsequent rates of range shifts, are not spatially homogenous (Chen, Hill, Ohlemüller, Roy, & Thomas, 2011; Sorte, Williams, & Carlton, 2010; Sunday et al., 2015; van Gennip et al., 2017). The magnitude and direction of climate velocities - a metric that describes both the speed and climate movement at any spatial location -, drive the patterns of range shifts in many marine species (Molinos et al., 2016). A climate velocity corresponds to the rate and direction in which a species would need to undergo a range shift to maintain its' current climate conditions. Areas where climate velocities converge and slow down and areas where rapid climate velocities terminate (i.e., climate sinks) concentrate range-extending species, often increasing tropical diversity in temperate areas, and novel species interactions (Burrows et al., 2014; Fogarty, Burrows, Pecl, Robinson, & Poloczanska, 2017). Most identified climate velocity convergence areas are in seas of sub-tropical latitudes, including south-eastern Australia, the North-eastern Atlantic, south-eastern Japan, and Chile. Many of the climate convergence areas identified by Burrows et al. (2014) align with the ocean warming hotspots identified by Hobday and Pecl (2014). They identified 24 regions experiencing ocean warming at rates faster than 90 % of the rest of the oceans and with unprecedented rates of species range shifts. The majority of these 'hotspots' of ocean warming were in the path of subtropical boundary currents, with the most significant warming and rate of shifting isotherms occurring proximal to Western Boundary Currents (Hobday & Pecl, 2014).

Western Boundary Currents form the western branches of large subtropical ocean gyres and transport warm tropical water poleward along the eastern coast of continents. Climate change is causing boundary currents to become stronger and extend further toward the poles resulting in unprecedented, localised warming in high latitude locations (Yang et al., 2016) and rapidly changing species' distributions (Sunday et al., 2015). For example, the strengthening of a Western Boundary Current, the EAC, has resulted in some of the fastest increases observed in ocean temperatures globally (Ridgway, 2007). Consequently, poleward range extensions to south-eastern Australia have been documented for 45 fishes representing approximately 15% of the coastal fish fauna and 16 intertidal invertebrates (Last et al., 2011; Pitt, Poloczanska, & Hobday, 2010).

Definitively attributing range shifts and biodiversity change to anthropogenic climate change is complicated by the interaction of other human factors and regular climatic variability. The interplay between cycles of climatic variability, including the Pacific Decadal Oscillation (PDO), Atlantic Multi-decadal Oscillation (AMO) and El Niño-Southern Oscillation (ENSO), drive regional ecology and have strong influences on global marine ecosystems (Hoegh-Guldberg et al., 2014). This natural seasonal, annual, and decadal variation in climate can influence the statistical inference of links between climate change and the responses of marine species at global and regional scales. For example, ENSO and PDO caused the Bering Sea to cycle between one of the warmest years in the past century to one of the coldest in the space of three years, causing widescale changes in plankton, fish and seabird communities (Batten & Walne, 2011; McKinnell & Dagg, 2010). Furthermore, stochastic events, such as heatwaves, can have major implications for predictive models of species distributions that are based largely on long-term or gradual warming trends (Wernberg et al., 2013). For example, a heatwave in Western Australia driven by an extreme La Niña event resulted in a significant reduction in the abundance of habitat-forming seaweeds and a subsequent shift in community structure towards low diversity and a tropicalisation of fish communities (Wernberg et al., 2013). This altered state mirrors the effects of climate-mediated warming but can be attributed to a rapid stochastic event. Other anthropogenic factors such as fishing pressure, eutrophication and habitat modification also complicate the detectability of ecological climate change climate signals by impacting species or population sensitivity to climate-mediated warming or intensifying impacts (Poloczanska et al., 2016; Pörtner & Peck, 2010). For example, climate change drove the poleward range shift of the tropical urchin *Centrostephanus rodgersii* to Tasmania; however, the intensive commercial and recreational fishing of its natural predator *Jasus edwardii* is likely to have increased its rate of establishment and population growth (Ling, Johnson, Frusher, & Ridgway, 2009).

Although it is now widely accepted that climate change in the marine environment is causing more frequent range shifts and that these range shifts can impact recipient ecosystems, they do not yet have the same notoriety as other stressors such as invasive species or overfishing (Johnson et al., 2011; Sorte et al., 2010). Several range shifts have already resulted in negative impacts on recipient ecosystems – the novel species have competed with native species (Arrontes, 2002) and modified community structure (Holbrook, Schmitt, & Stephens, 1997; Ling, 2008; Madin et al., 2012) – resulting in environmental, economic and social challenges (Johnson, Ling, Ross, Shepherd, & Miller, 2005; Pecl et al., 2017). In a study comparing the relative impacts of range shifting and invasive species, Sorte et al. (2010) found that they only had evidence of ecosystem-level impacts for a small portion of the range shifting species they studied (8 of 129). However, the impacts of range shifting species on recipient ecosystems were diverse in both type and intensity. For example, increased competition, predation, herbivory, and disease incidence resulted in a 13% decrease in microalgal biomass on oyster shell substratum due to grazing by a range shifting crab (*Petrolisthes armatus*). Arguably one of the best-documented impacts of range shifts is that of a phase shift from productive kelp-dominated ecosystems to urchin barrens due to grazing by the range shifting sea urchin *C. rodgersii* in Tasmania (Ling et al., 2009). Long-term warming and grazing by herbivorous range-extending fishes have also been implicated in a novel phase-shift away from macroalgal dominance to barrens and coral dominated ecosystems in the Mediterranean Sea and Japan (Kumagai et al., 2018; Vergés et al., 2014).

1.4 Ocean climate change in New Zealand

New Zealand (NZ) is an island nation situated over 1600km east of Australia and 1000km south of New Caledonia, Fiji, and Tonga. NZ consists of three main islands that span 1500km across 13 degrees of latitude from 34°S to 47°S, and the ~18000km coastline is one of the longest in the world. New Zealand’s coast is further complicated by more than 700 offshore islands and rock stacks, most of which are located within 50km of the coast (Gordon, Beaumont, MacDiarmid, Robertson, & Ahyong, 2010). About two-thirds of the coast consists of hard rocky reefs, with soft-sediment types including sand and gravel making up the remainder. NZ’s wide latitudinal range generates a strong north-to-south cline in sea surface temperature. Monthly mean Sea Surface Temperatures (SST) ranges from 18-24°C at the Rangitāhua/Kermadec Islands in the north (Francis et al. 1987; Francis 1993) to 3-7°C at Campbell Island south of mainland NZ (Williams 1988). Not surprisingly, the geographical distributions of the region’s marine plants (Moore 1961; W. Nelson 1994) and animals (Moreland 1959; Knox 1963; Pawson 1965; Dell 1968; Francis 1993; Paulin & Roberts 1993) are strongly influenced by this variation in the oceanographic environment. NZ’s oceanographic setting, latitudinal spread, and lengthy coastline have resulted in a wide variety of habitats and high species diversity relative to other isolated island nations and temperate locations (Roberts, Stewart, & Struthers, 2015).

Some of the highest diversity of marine organisms in NZ are associated with the warm, subtropical waters in north-eastern NZ from North Cape to East Cape (Gordon et al., 2010). North-eastern NZ is a convergence area for a warm subtropical current, the East Auckland Current (EAUC). At approximately 33°S, a branch of the EAC splits east away from mainland Australia and forms the Tasman Front, this warm, slow-moving front moves east from Australia past Lord Howe and Norfolk Island before wrapping around Cape Reinga as the EAUC (Laing, Stanton, & Challenor, 1996; Stanton, 1981). The warm EAUC moves Southeast along the continental shelf between North Cape and East Cape, where it acts to moderate the sea surface temperatures, resulting in waters off north-eastern

NZ being around 1°C warmer than the oceans off the west coast of NZ at similar latitudes (Sutton & Roemmich, 2001).

The oceans around NZ are warming but the observed and predicted locations, rate and intensity of climate change are highly variable. Research suggests that significant long-term warming is occurring in areas south of the East Cape biogeographic break, particularly the Wairarapa (Sutton & Bowen, 2019) and Dunedin (Shears & Bowen, 2017), with SST's increasing at rates of up to 3.5°C per decade (Shears & Bowen, 2017). However, short- and long-term ocean temperature trends display significant spatio-temporal variability. The most substantial interannual variability in NZ SST is attributed to the El Niño/Southern Oscillation (ENSO) (e.g., Greig et al. 1988; Folland and Salinger 1995; Sutton and Roemmich 2001; Bowen et al. 2017). Historically cooler annual temperatures are associated with El Niño (negative SOI), and warmer temperatures positively correlated with La Niña dominated years. At more localised spatial scales, trajectory, and proximity of the EAUC to the coast impacts local sea surface temperatures (Sharples, 1997; Stevens et al., 2019; Stanton & Sutton, 2003; Zeldis, Oldman, Ballara, & Richards, 2011). In La Niña years, north-easterly winds drive the warm waters into nearshore environments, and coastal water temperatures increase; during El Niño years, the opposite occurs, south-westerly winds prevail, and these push the EAUC further offshore, leading to lower inshore water temperatures (Bradford-Grieve, Chang, Manning, Paul, & Tasker, 2005; Shears & Bowen, 2017). Furthermore, ocean warming trends in shallow reef systems around NZ vary according to local weather patterns. Storm events, localised solar heating and wind conditions directly impact SST in shallow coastal waters and drive the rate of upwelling and the location of large current systems like the EAUC (Sutton, Bowen, and Roemmich, 2005; Bowen et al., 2017). These complex environmental and oceanographic factors contribute to the variability in ocean warming trends and the rate and location of subsequent poleward range extensions in NZ (Sutton & Bowen, 2019).

NZ’s marine environment and biodiversity has remained relatively unimpacted by climate change compared to areas at similar latitudes (e.g., Johnson et al. 2011; Ridgway and Hill 2012). At present, there are few cases of clearly documented climate-change impacts in NZ’s coastal marine environment. The most compelling evidence is from heatwave impact research; significant losses of large kelp species, including Bull kelp, *Durvillea poha* (Thomsen et al., 2019) and Giant kelp, *Macrocystis pyrifera* (Tait, Thoral, Pinkerton, Thomsen, & Schiel, 2021) have been attributed to ocean heatwaves. Furthermore, several recreationally, commercially, and culturally significant fish and invertebrate species also show signs of climate-mediated impacts. For example, the reproductive phenology of the sea urchin (*Evechinus chloroticus*) has changed due to ocean warming and the distribution and abundance of commercially important fish species such as Snapper (*Pagrus auratus*) and Kingfish (*Seriola lalandi*) have shifted due to long term ocean warming and marine heatwaves (Ministry for the Environment & StatsNZ, 2019). However, empirical evidence of climate-mediated impacts and range shifts in NZ’s marine environment is limited due to the limited temporal scale of existing studies, a current lack of significant or measurable effects (Macinnis-Ng et al., 2021), and the complex oceanographic setting (Hewitt, Bulmer, Stephenson, & Thrush, 2021; Lundquist, Ramsay, Bell, Swales, & Kerr, 2011).

Tropical and subtropical fishes have been recorded sporadically in NZ waters over the last 100 years (Roberts, Stewart, & Struthers, 2015) and could serve as indicator taxa for biodiversity responses to climate change. Pulses of larval dispersal and adult migration of tropical and subtropical fishes to NZ are anecdotally associated with nearshore sea surface temperature anomalies which result from long term variations in the El Niño Southern Oscillation (ENSO; Francis & Randall, 1993; Francis, Worthington, Saul, & Clements, 1999), variability in the strength and location of the EAUC (Laing, Stanton, & Challenor, 1996; Stanton & Sutton, 2003) and local weather patterns [Bowen et al. (2017); Sutton et al. (2005)]. Most of the tropical and subtropical fishes that arrive in NZ are ephemeral, and they consistently undergo significant population declines during Austral winters. Globally a threshold of between 17 and 18°C has been identified as the temperature below which few tropical reef fishes can survive (Booth, Beretta, Brown, & Figueira, 2018; Figueira & Booth, 2010). The average sea surface temperature in the warmest parts of NZ is around 17°C, but winter temperatures drop to ~15°C (Philip Sutton & Roemmich, 2001), well below the recognised threshold for tropical reef fish survival. However, NZ ocean temperatures are expected to increase at a rate of up to 3.5°C per century, and most significantly, winter minima are increasing at rates of up to 0.3°C per decade (Shears & Bowen, 2017). Under present ocean-warming predictions, areas in NZ could reach winter temperature thresholds for more cold-tolerant tropical fishes within the next 50 years.

The tropical and subtropical fishes most likely to persist long term at high latitude locations like NZ are dietary generalists with broad environmental tolerances and large body sizes at settlement and maturity (Feary et al., 2013). For example, NZ reefs are largely devoid of reef-building corals (Grigg, 1982; Muir, Wallace, Done, & Aguirre, 2015), and we can expect that families with high proportions of coral dependent species, such as Chaetodontidae, Apogonidae and Gobiidae, are less likely to contribute to range shifts. In contrast, herbivorous or omnivorous range shifting species are likely to be capable of exploiting novel food sources in NZ's macroalgal dominated temperate reef ecosystems if the thermal regimes do not exceed critical thresholds (Basford et al., 2016; Vergés et al., 2014).

Like many isolated island nations, the NZ fish fauna exhibits a high level of endemism. Endemic species are more prone to localised and broad-scale climate-mediated population declines; even the slightest environmental and temperature shifts may be outside of the narrow environmental tolerances of endemic species. Range extending generalist species could exhibit competitive dominance over local species, be more effective at avoiding novel predators and be more capable of exploiting novel food sources and habitats (Bates et al., 2014; Sunday et al., 2015, Luiz et al., 2012). Novel species interactions could lead to losses of local specialist or endemic species in favour of range-shifting generalists (Monaco et al., 2020), which in turn could lead to the loss of essential and highly specialist ecosystem functions (Munday, 2004).

Even though NZ has a lot to lose if new species extend their ranges or endemic species undergo population declines, there has been little effort to consolidate the records of new to NZ fishes - those fishes observed outside of their known range - or monitor the spatio-temporal patterns of rare, tropical, and subtropical fish populations in NZ waters. Although not considered a hotspot of biodiversity change like Southeastern Australia (Hobday & Pecl, 2014; Shears & Bowen, 2017), it is unlikely that our oceans will remain immune to the impacts of climate change and range shifting species. The spatio-temporal variability in ocean warming trends, coastal complexity and NZ's oceanic isolation suggest that range shifts and biodiversity change in NZ are likely to consist of a mosaic of continued species arrivals, increased survival of cold-tolerant and generalist species, changes in the seasonal occurrence of tropical fishes and poleward range shifts of native species (Bradford-Grieve, Chang, Manning, Paul, & Tasker, 2005; Francis & Duffy, 2015; Francis et al., 1999). Although subtle, these early indicators of biodiversity change in coastal marine environments forewarn more significant biodiversity impacts.

1.5 Monitoring range shifts

Early detection of novel and range shifting species can limit potential impacts, allow for the recognition of potential source populations, and identify new potential fisheries or commercially important species. However, at the early stages of establishment, range shifting species are inherently rare, and the accurate and timely identification of range shifts requires high-resolution data at broad spatio-temporal scales. Although technological advances have led to significant increases in the availability of broad-scale biological data and the development of large online databases such as the Global Biodiversity Information Facility (https://www.gbif.org/) and Ocean Biogeographic Information System (https://obis.org/), these data still have significant spatial biases. They often lack quantitative abundance information that is particularly important for monitoring range shifts. Furthermore, systematic surveys at the scale required to identify range shifts are resource-heavy and expensive and so are often restricted to specific high value or high-risk areas and species.

At the early stages of range shifts, stochastic range dynamics, including long-distance dispersal and migration, can confound the accurate identification of range shifts and the underlying drivers (Bates et al., 2014; Robinson et al., 2015). For example, storm events may advect fishes, plants, or birds outside their natural range; however, many may not persist long-term. Monitoring at appropriate temporal scales will help to distinguish these stochastic arrivals from those species that can persist long term, representing actual range shifts. Range shifts also do not always follow the simple trajectory of larvae dispersing to a new location, surviving, and persisting to the breeding stage (Bates et al., 2014). Range shifts can occur for different life-history stages, such as larvae or mobile adults, and new populations may represent self-recruiting populations or require sustained long-distance larval dispersal from distant source populations (Bates et al., 2014). Furthermore, the biotic and abiotic factors setting range limits differ among species and are underpinned by physiological and demographic processes (Lenoir and Svenning, 2013), resulting in high levels of interspecific variability in the rates of range shifts (Brown et al., 1996, Gaston, 2003, Sexton et al., 2009, Doak and Morris, 2010, Bates et al., 2014). The level of detail required to identify range shifts and accurately account for the background 'noise' associated with natural range dynamics remains elusive for many species, particularly in the marine environment. Large areas of the marine realm remain under-surveyed (Hortal et al., 2015), and marine species are often cryptic (Hubert et al., 2012), wide-ranging, and highly mobile (e.g., Block et al., 2005; Bonfil et al., 2005), so our knowledge of geographic distributions is far from complete (Appeltans et al., 2012; Hortal et al., 2015).

The marine environment is challenging and expensive to observe, and it is impossible to monitor the entire ranges of all potentially shifting species. Hence, marine scientists and managers must often rely on predictions, models, and diverse sources of knowledge to identify range shifts

(Delaney, Sperling, Adams, & Leung, 2008). Recently, a number of studies have mined existing datasets such as fisheries data (e.g., Robinson et al., 2015), museum collections (e.g., Cheung, Watson, & Pauly, 2013; Perry, Low, Ellis, & Reynolds, 2005) and opportunistic species occurrence records provided by the general public (e.g., Lenanton, Dowling, Smith, Fairclough, & Jackson, 2017; Liggins et al., 2020) to augment existing knowledge of species range dynamics and identify range shifts. Recent technological advances have also increased the availability of largescale databases such as online biodiversity information repositories sourced from peer-reviewed sources and the public (e.g., GBIF; OBIS; and iNaturalist [https://www.inaturalist.org/]). Increasingly, citizen science – the participation of citizens in the collection of data or the provisioning of local knowledge for the planning, development and execution of scientific research (Eitzel et al., 2017) – is leading to increased temporal and spatial data resolution, overcoming a lack of baseline data and detecting out-of-range occurrences in both terrestrial (Theobald et al., 2015; e.g., The Great British Birdcount, MonarchWatch.org) and marine settings (Pecl et al., 2019; e.g., Range Extension Database and Mapping Project [Redmap] Reef Environmental Education Foundation [REEF]; European Alien Species Information Network [EASIN]). However, even though citizen science data has proven utility in contributing toward scientific (e.g., Lenanton et al., 2017; Soroye, Ahmed, & Kerr, 2018) and policy outcomes (e.g., Delaney et al., 2008; Madin et al., 2012), it is often underutilised given perceived biases and lack of quality (Bird et al., 2014; Isaac, Strien, August, Zeeuw, & Roy, 2014).

1.6 Utility of citizen science in range shift monitoring

Citizen science programmes have a number of direct and indirect benefits; they can play a valuable role in increasing engagement between the public and science and are essential to developing large-scale, high resolution global datasets. The spatiotemporal scale of citizen science projects makes them particularly well suited to the fields of range shift detection and invasive species monitoring, where species are often rare or patchily distributed. Leveraging a pool of interested and invested citizens allows for the development of large-scale, long-term datasets that can be constructed at a low cost relative to employing professionals for the same purpose. In particular, citizen science can increase species distributional data resolution and fill recognised knowledge gaps, increasing the utility of these data to identify climate change biodiversity impacts, including poleward range shifts (Devictor, Whittaker, & Beltrame, 2010; Mieszkowska et al., 2006; Parmesan & Yohe, 2003). For example, Mieszkowska et al. (2014) identified the fastest range shifts in marine species in 50 years using citizen science data collected as part of the Marine Biodiversity and Climate Change (MarClim) project. Furthermore, citizen science projects generally engage the local or affected populations throughout the stages of hypothesis setting, experimental design, and data collection. This high level of involvement results in social outcomes and benefits, including improved scientific and ecological literacy, enhanced stewardship, and a stronger sense of community and shared purpose (Dickinson et al., 2012; Thiel et al., 2014).

Although citizen science is not a new concept, research involving public participation has increased significantly in the last 20 years (Silvertown, 2009; Thiel et al., 2014; Theobald et al., 2015). The first and most long-standing recognised citizen science project, the National Audubon Society Christmas Bird Count, was established in 1900 (Cohn, 2008), and 60,000 to 80,000 participants are currently still involved in this project. At present, there are millions of participants involved in citizen science projects that range in focus from the deep ocean to the solar system and in scope from opportunistic observations, environmental monitoring to the timing of species phenological patterns (Dickinson et al., 2012). The design of citizen science projects and the level of citizen participation can be classified along a spectrum (Shirk, Ballard, Jordan, & Bonney, 2011, Shirk et al., 2012) from non-targeted contributory programs that involve a large number of contributors, with little or no training (e.g., iNaturalist), to more focussed opportunistic observation projects that set a priori research questions or hypotheses (e.g., Redmap), or projects that have a small number of highly trained contributors (e.g., Reef Life Survey) where the accuracy of the data is similar to data collected by professional scientists (Edgar and Stuart-Smith, 2009). Contributory models of citizen science (e.g., low training, collection of observations) have been the most productive in generating large broad-scale datasets that have been used regularly in peer-reviewed publications. In contrast, projects involving collaboration between citizens and scientists (e.g., citizens involved in hypothesis setting and project design or implementation) are often more productive for developing practical management and conservation projects (Dickinson et al., 2012).

Although not as prevalent as their terrestrial counterparts (Roy et al. 2012; Cigliano et al. 2015; Theobald et al. 2015; Garcia-Soto et al. 2017), marine citizen science projects have increased in popularity and provide a valuable tool for enhancing scientific monitoring, modelling and observational power in a demanding environment (Bonney et al. 2009; Silvertown 2009; Hochachka et al. 2012; Garcia-Soto et al. 2017). However, the logistical difficulties related to subtidal and deep-water sampling, means that most citizen science research is concentrated in easily accessible habitats, including intertidal habitats, estuaries, beaches, seagrass beds and shallow coral reefs (Thiel et al., 2014). Furthermore, the focal species for most studies reflect citizen scientist motivations and interests and are often skewed toward commercially important, charismatic, and endangered species (Earp & Liconti, 2020; Thiel et al., 2014). Actinopterygii, the ray-finned fishes, are the most common focal species for marine citizen science projects (Thiel et al., 2014). The most well-known and longest-standing fish survey project is the REEF (Reef Environmental Education Foundation) Fish Survey Project (http://www.reef.org), a monitoring programme that engages volunteer divers and snorkelers to survey species diversity and abundance using a standardised survey technique. The REEF Fish Survey Project started in Florida in 1993 and was consequently promulgated worldwide; by 2020, the database included more than 250,000 surveys collected by over 16,000 citizen science divers and snorkellers (Pattengill-Semmens & Semmens, 2003).

Surveys of invasive species and range shifting species in the marine environment have successfully taken advantage of the fact that citizen scientists are often spread across areas that exceed jurisdictional and national borders. For example, in response to nearby incursions of the globally recognised pest, the European green shore crab (*Carcinus maenas*), a citizen science-based early detection and monitoring project, was identified as the most effective method to surveil the nearly 3,000 km of Washington's inland shoreline (Grason et al., 2018). Citizen science volunteers monitored 26 sites throughout multiple counties and successfully detected the first incursion of this pest to the area (Grason et al., 2018). In Tasmania, continued rapid ocean warming and anecdotal observations of range shifting species drove the development of the Range Extension Database Mapping Project (RedMap). RedMap is a citizen science project that engages with divers, fishers, and boaters to collect sightings of species that are likely to be tracking isotherms poleward. The project expanded nationally across Australia at the end of 2012, and by 2019 over 2,000 observations of focal species were submitted by 784 citizen scientists (Pecl et al., 2019). RedMap data has been used in several peer-reviewed publications that have significantly increased the understanding of climate change-driven species and ecosystem changes occurring in southeast Australia, a globally significant ocean warming hotspot (Hobday and Pecl, 2014).

In NZ, citizen science is recognised as an essential data collection method to enhance the public understanding of science and strengthen links between professional scientists and community members (Ministry for the Environment & StatsNZ, 2019). Recent policy directions include reference to the importance of citizen science for environmental monitoring and increased engagement and community ownership. For example, Biosecurity NZ's 2025 vision has set out five strategic directions, one of which is to make all NZ citizens aware of the importance of biosecurity and to get them involved in pest and disease management by encouraging a collective effort across the country – in which 'every New Zealander becomes a biosecurity risk manager, and every business manages their own biosecurity risk'. The Ministry of Business, Innovation and Employment, the Ministry of Education and the Office of Prime Minister's Chief Science Advisor also developed 'Curious minds-He hihiri I te mahara', an initiative that has funded the development of 242 citizen science projects throughout NZ, including 32 marine-based projects (https://www.curiousminds.nz/projects/). Currently, NZ's citizen science programmes range from national (such as the bird counts coordinated by Birds NZ) to local, site-specific initiatives (such as those coordinated by community environmental groups and via the Curious minds platform).

Many of the NZ citizen science projects use the web-based application iNaturalist to log data and make it available to a broader audience. Therefore, iNaturalist outputs can be used as a proxy for the environmental monitoring impact of citizen scientists (Peters, 2018). To date, iNaturalist had secured over 1 million observations in NZ with more than 16,300 identified species from 21,700 observers. Most of the NZ citizen science projects registered in iNaturalist are in terrestrial environments, and the fact that ray-finned fishes only comprise 1.1% of iNaturalist observations (https://inaturalist.nz/) suggests that the potential for marine citizen science has not yet fully been explored. The citizen science projects that are based in NZ's marine environment are often limited to single species or highly localised sites (e.g., South Taranaki Reef Life Project [https://www.projectreefsouthtaranaki.org/], Finding little blue [https://www.findinglittleblue.nz/about/]). One of the few marine citizen science projects in NZ that operates on a national scale is the Marine Meter Squared project (MM2), an intertidal monitoring project that uses set methodology to monitor marine diversity and change within a 1m quadrat (https://www.mm2.net.nz/). MM2 currently has 2683 registered users spread from Northland through to Southland. Although the MM2 project has recently added marine invasive species to their focal species, citizen science projects monitoring climate-mediated range shifts at local or national levels are lacking in NZ's terrestrial and marine environments.

Overseas studies like RedMap and Reef Life Survey have proven the utility of marine citizen science observations to identify range shifting marine species and monitor biological and environmental drivers and ecological impacts of climate change. Even though there is a vast amount of knowledge and observational power available to NZ scientists in the form of fishers, divers and spearfishers, their informal records of potentially range shifting fishes or ecosystem change have not been harnessed. The last published study that included citizen science observations of rare, tropical, and subtropical fish species in NZ waters was by Francis et al., (1999). Many of the 25 new and rare tropical fishes were photographed and observed by fishers and scuba divers. Novel species introductions and range shifting species are set to increase in NZ waters (Francis et al, 1999; Shears

& Bowen, 2017), but they will be difficult to detect as it is likely that they will be diffusely distributed, and present as a mosaic of new species arrivals or increased persistence. Citizen science projects that harness the observational power of NZ ocean-going citizens by capturing opportunistic sightings of focal species at broad spatial scales will represent the most efficient monitoring option for range shifts in NZ. Citizen science in NZ would not only have the potential to identify range shifting marine species, but also increase the awareness of climate-mediated biodiversity change, assist in streamlining policy and management outcomes, and increase the general public's engagement with science outcomes.

Evidence suggests that citizen science data can provide information at similar qualities to professionally collected and designed monitoring programs (Delaney, Sperling, Adams, & Leung, 2008; Szabo, Fuller, & Possingham, 2012). However, many projects still encounter issues relating to data completeness and objectivity (Bird et al., 2014; Isaac, Strien, August, Zeeuw, & Roy, 2014). The biases encountered in citizen science projects often present as spatial and temporal clustering due to the behaviour of the observers. For example, divers, spearfishers and fishers often frequent locations with high diversity resulting in over-representation of these sites in datasets and difficulty identifying real trends. Temporal biases in citizen science data collection can result from weather conditions, societal change, and technical advances such as the availability of camera equipment, smartphones and the popularity of scuba diving (Pecl et al., 2019). As a consequence, increased frequency of observations can be easily misinterpreted for increased density of target species and needs to be accounted for in data analysis or interpretation.

Various a priori and post hoc methods have been proposed and tested to minimise sampling biases and maximise data quality in citizen science projects. Delaney et al., (2008) demonstrated that the accuracy of invasive crab identification increased between third grade (80% accurate) and seventh grade (95% accurate) high school students, suggesting that demographic variables such as age and educational background may be important drivers of data quality. Other studies suggest that the training of participants by project coordinators and participating scientists significantly increased data quality (Edgar & Stuart-Smith, 2009). Furthermore, collecting, and analysing data at appropriate resolution can minimise data biases and inaccuracies. If there is sufficient data and a high level of consensus when analysed at the appropriate level, data of perceived lower quality to due spatial or temporal observation biases or inaccuracies can provide high confidence in trends (Mastrandrea et al., 2010). For example, examining trends at a family level rather than a species-level or regional rather than local scales may result in more accurate analyses, particularly when data and resources are limited (Gouraguine et al., 2019). Alternatively, project coordinators and scientists can encourage unrestricted broad scale, high-resolution data collection by citizen participants, knowing that there may be data errors or biases. In these situations, post hoc data validation and verification methods should be included in the project protocol (Tweddle, Robinson, Pocock, & Roy, 2012). Despite the importance of being able to defend the quality and credibility of citizen science data to the broader scientific community, only half of the marine citizen science projects in a 2014 review included data validation or verification protocols (Thiel et al., 2014). However, more recent advances in the training protocols for citizen scientists [e.g., Edgar and Stuart-Smith (2009)], the development of validation tools that incorporate expert knowledge [e.g., Pecl et al. (2019) & Robinson et al. (2015)] and the increased availability of statistical options for handling common biases [e.g., Bird et al. (2014)] have paved the way to making volunteer data more accepted by the scientific community.

Although demonstrated successes are increasing, marine citizen science still suffers from unrealised potential due to its novelty and continued challenges (Thiel et al., 2014). To meet its full potential, citizen science practitioners must work cooperatively to acknowledge common shortcomings and develop novel tools and methods to ensure the generation of high-quality data sets (Burgess et al., 2017). The ‘many eyes’ aspect of citizen science is particularly valuable for monitoring rare and range shifting species. As climate change continues to result in marine biodiversity change, habitat degradation and species losses, it is likely that scientists will increasingly look toward citizen science to fill knowledge gaps, increase awareness, and address global challenges. NZ's oceans are not immune to climate-mediated impacts, and despite the continued lack of scientific resources, our ocean-going citizens' incredible observational power remains largely underutilised. There is significant scope for the development of citizen science projects, validation and verification protocols and engagement that will allow for the incorporation of 'science by the people' into marine research and conservation projects in our rapidly changing oceans (Earp & Liconti, 2020; Thiel et al., 2014).

1.7 Thesis outline

I use tropical, subtropical, and rare marine teleost fishes as tohu, or signals, of change in NZ waters. Teleost fishes are early responders to climate-mediated ocean warming and are valuable indicators of current and future change. Teleost fishes are also ideal focal species for citizen science projects as they are of high customary, recreational and commercial value, and can be highly visible and charismatic. The primary aims of this thesis were to set an accurate baseline for the spatio-temporal distribution of tropical, subtropical, and rare fishes in NZ waters and develop methods to allow us to monitor future biodiversity change.

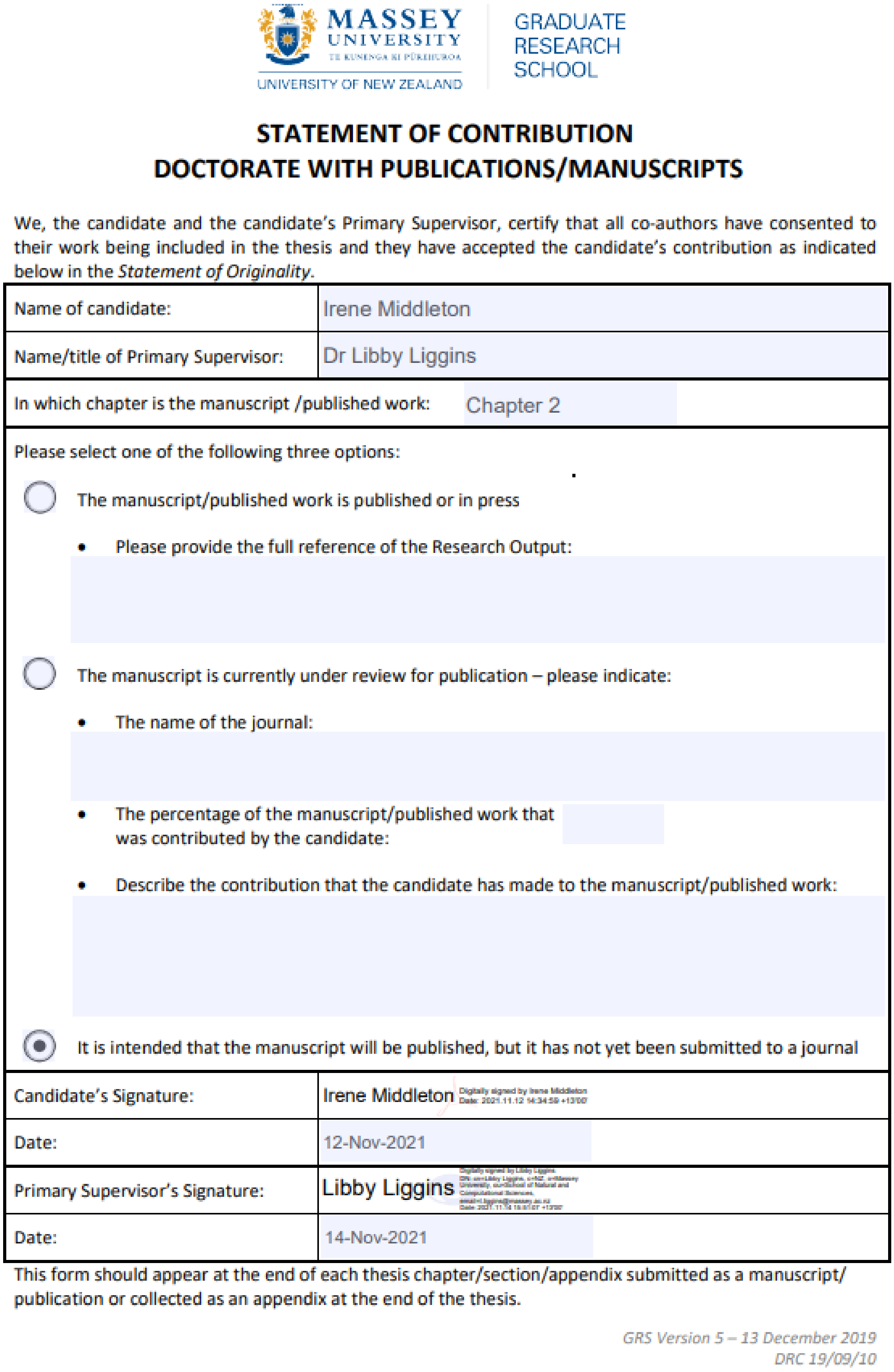
The thesis contains three empirical data chapters (Chapters Two, Three, and Four) and a general discussion chapter (Chapter Five). Chapters Two-Four have been prepared as peer-reviewed papers for publication; hence there is some repetition of methodological detail throughout the thesis, particularly for Chapters Three and Four. As these papers are co-authored, throughout, I use "we" or "our" (first-person plural) to acknowledge this (see "Declaration by Author" on page IV, and the DRC16 Statement of Contribution forms at the end of each chapter for details on authorship contributions). Chapter Two and Three are in preparation for publication in the Journal of Biogeography and Global Change Biology respectively. Chapter Three has been published in the journal Diversity and Distributions.

In **Chapter Two**, we undertake the first dedicated analysis of tropical, subtropical, and rare fish distributions in NZ since 1999. We examine spatio-temporal patterns in the occurrences of tropical, subtropical, and rare marine fishes in temperate NZ to characterise their biodiversity contribution and to set a baseline for future monitoring of climate change impacts. We derive most of the observations from citizen science sources and present 13 fish species and one family not previously recorded in mainland NZ and four fishes that had been recorded at Rangitāhua/Kermadec Islands previously but were new-to-NZ mainland. This chapter demonstrates the utility of citizen science for biodiversity monitoring in NZ and that tropical, subtropical, and rare fishes constitute a significant component of the fish biodiversity in temperate NZ and that their occurrences and diversity have increased over the past 50 years. The co-authors are currently reviewing this chapter for publication in the Journal of Biogeography.

Chapter Three describes a novel approach that combines citizen science with expert knowledge to classify out-of-range occurrences for marine fishes as potential range extensions or human-mediated dispersal events. The stepwise approach includes decision trees, scoring, and matrices to classify citizen science observations of species occurrences and provide a measure of confidence and validation using expert knowledge. We apply this method to a subset of 10 tropical, subtropical or rare fishes to demonstrate its utility and broad scope of applications, including marine biosecurity, climate change impact assessments and conservation. By applying the novel method, we identify that four of the ten test species had potentially extended their ranges into or within NZ waters, and one species occurrence was likely due to human-mediated dispersal.

Chapter Four comprehensively identifies which subtropical, tropical, and rare fishes have undergone range shifts to and within NZ waters and where biodiversity change is occurring in NZ as a result. We apply the method described in Chapter Three to the entire database of citizen science sourced occurrence data for tropical, subtropical and rare fishes to identify range shifts. Range shifts are classified as occurrences of mature individuals of focal species outside of their known range. We then use hotpot analyses to identify the different taxa that are contributing disproportionately to range shifts in NZ waters and the regions in which there are significantly more range shifts are occurring. We analyse spatial trends in out-of-range species by decade, season, habitat, and life stage to develop an accurate list of species contributing to range shifts by region. These lists could be used by resource managers, science practitioners and the public as early indicators of broadscale biodiversity shifts. This chapter further demonstrates the broad utility of the classification tools described in Chapter Three and how citizen science data can streamline marine resource management and conservation.

Chapter Five discusses the significant findings across all chapters and draws conclusions based on the thesis as a whole. I also present future directions and potential research questions that have arisen as a result of this work. Author contributions are indicated in each chapter.



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STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the candidate and the candidate's Primary Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of candidate:	Irene Middleton
Name/title of Primary Supervisor:	Dr Libby Liggins
In which chapter is the manuscript /published work:	Chapter 2

Please select one of the following three options:

○ The manuscript/published work is published or in press

- Please provide the full reference of the Research Output:

○ The manuscript is currently under review for publication – please indicate:

- The name of the journal:
- The percentage of the manuscript/published work that was contributed by the candidate:
- Describe the contribution that the candidate has made to the manuscript/published work:

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Candidate's Signature:	Irene Middleton Digitally signed by Irene Middleton Date: 2021.11.12 14:34:59 +13'00'
Date:	12-Nov-2021
Primary Supervisor's Signature:	Libby Liggins
Date:	14-Nov-2021

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GRS Version 5 – 13 December 2019
DRC 19/09/10

2 Indicators of change: spatio-temporal variability in the occurrences of tropical, subtropical and rare marine fishes in Aotearoa New Zealand.

2.1 *Abstract*

Aim Climate change is driving biogeographic change globally, including poleward range shifts of species and the increasing abundance of rare species. We examine spatio-temporal patterns in the occurrences of tropical, subtropical and rare marine fishes in temperate New Zealand, with the purpose of characterising biodiversity change and to set a baseline for future monitoring of climate change impacts.

Methods We consolidated 50 years of unpublished records and citizen science sightings of tropical, subtropical, and rare fishes to develop a focal species database for NZ. Using hotspot analysis, we identify geographic locations where the occurrence and diversity of focal species is greater than expected. We examine the spatio-temporal variation in hotspots in relation to focal species life-stage, habitat, and taxonomic family.

Results The diversity and occurrences of focal species in NZ have increased. We present 17 new-to-NZ marine fish species, and a new-to-NZ family. Focal species now account for 6.5% of known pelagic fishes and 17.3% of all benthic fishes in NZ. North-eastern NZ was a consistent hotspot of focal species occurrences and diversity. Differences were subtle, but hotspots of mature and pelagic fish occurrences appeared further south than juvenile and benthic species occurrences. Families with tropical affinities (Labridae and Pomacentridae) were restricted to north-eastern NZ whereas hotspots of families with more temperate affinities (i.e., Cheilodactylidae), extended further south.

Main conclusions Tropical, subtropical and rare fishes are a major component of the fish biodiversity in temperate NZ, and their occurrences and diversity has increased over the past half-century. North-eastern NZ consistently has the highest occurrence rate and diversity of warmer water fishes, and the patterns of species occurrences in this area could inform future biodiversity

shifts around NZ. Our study demonstrates the value in consolidating citizen scientist observations to inform current baselines and future monitoring of climate-related biodiversity change.

2.2 *Introduction*

Climate change is causing the distinctness of species assemblages and biogeographic regions to be eroded (Davey, Chamberlain, Newson, Noble, & Johnston, 2012; Poloczanska et al., 2013). The expectation is that as global temperatures increase, tropical regions will become increasingly inhospitable, and temperate regions will become more tropicalised and biodiverse as species track favourable conditions poleward (Molinos et al., 2016). Accordingly, studies have reported increases in tropical plant (Root et al., 2003), butterfly (Parmesan et al., 1999), fish (Hiddink & ter Hofstede, 2008) and bird (H-Acevedo & Currie, 2003; van der Wal et al., 2013) diversity in high latitude regions linked to climate-driven poleward range shifts (Hickling, Roy, Hill, Fox, & Thomas, 2006). Though more subtle, greater increases in abundance at the poleward extremes of a species range relative to the core of a species range have also been observed as a consequence of climate change (Last et al., 2011). These occurrences of new species and the increased relative abundance of tropical species in high latitude locations, are considered early indicators of long-term biodiversity change in response to an altered climate (Last et al., 2011).

The most significant local biodiversity changes over recent decades have occurred in the coastal marine environments (Burrows et al., 2011; Poloczanska et al., 2013). Most marine organisms are ectotherms, making them highly sensitive to changes in local climate (Heath et al., 2012; Pinsky, Selden, & Kitchel, 2020; Pörtner & Peck, 2010; Sunday, Bates, & Dulvy, 2012). Furthermore, many marine species are highly mobile, either as pelagic larvae, as adults or both, increasing their ability to avoid suboptimal environmental changes by dispersing to new, more suitable, areas (Pinsky et al., 2020; Pinsky, Worm, Fogarty, Sarmiento, & Levin, 2013). Consequently, marine range shifts are occurring approximately four times faster than observed rates on land (Burrows et al., 2011; Chen, Hill, Ohlemüller, Roy, & Thomas, 2011; Poloczanska et al., 2013; Sorte, Williams, & Carlton, 2010).

Marine range shifts are stagewise; first, a species needs to immigrate to a new location, then persist in recipient environments and eventually breed and establish populations (Bates et al., 2014). These stages are differentially influenced by climate change. For instance, the rate of new species immigration or arrival events can be altered by climate-induced change in the strength and directionality of ocean currents (Booth, Figueira, Gregson, Brown, & Beretta, 2007); whereas changes in the local marine environment (e.g., sea surface temperatures, productivity) and altered seasonality resulting from climate change can impact the persistence and breeding of new species or previously rare species (Bates et al., 2014; Monaco et al., 2020). Thus, observable marine biodiversity changes may include new species arrivals, increased immigration of certain species, increased survival, maturity, or changes in their seasonal occurrence (Poloczanska et al., 2016; Sorte et al., 2010). Although subtle, these early and localised indicators of biodiversity change in a coastal marine environment forewarn more significant biodiversity impacts and provide insight as to the specific physical and environmental changes underpinning them.

Unlike many coastal marine ecosystems at similar latitudes, broadscale impacts of climate change have not yet been observed in Aotearoa New Zealand (NZ). NZ is a temperate, oceanic, island nation with a complex coastline and nearshore oceanography (Gordon, Beaumont, MacDiarmid, Robertson, & Ahyong, 2010, Figure 1). Consequently, patterns of marine biodiversity change and range shifts both to, and within, NZ may not follow patterns observed in other temperate regions (e.g., Ling et al., 2015; Mieszkowska et al., 2006; Sanford, Sones, García-Reyes, Goddard, & Largier, 2019). For instance, NZ is surrounded by an open ocean dispersal barrier; for coastal species to successfully disperse to NZ, they must disperse over 1000km of open ocean. Furthermore, the East Auckland Current (EAUC), the primary warm-water current implicated in the transport of immigrants from the tropics and subtropics, varies in strength and distance offshore (Sharples, 1997; Stanton & Sutton, 2003; Zeldis, Oldman, Ballara, & Richards, 2011) and complex coastal features disturb the major current flow creating variable eddy systems (Stevens, O'Callaghan, Chiswell, & Hadfield, 2019). Although coastal NZ has experienced ocean warming over recent decades (Shears & Bowen, 2017; Sutton & Bowen, 2019), and the frequency of marine heatwaves is set to increase (Behrens, Fernandez, & Sutton, 2019; Chiswell & Sutton, 2020), local weather patterns exert a strong influence on the overall trend [i.e., solar heating, and wind conditions; Sutton, Bowen, and Roemmich (2005) and Bowen et al. (2017)]. Detecting early biodiversity changes in NZ's marine coastal environment is vital but may be challenging given the lack of comparable biodiversity settings to predict outcomes of climate change for NZ's marine biodiversity.

One of the best described marine taxa in NZ are the ray-finned fishes, Actinopterygii (Gordon et al., 2010). Records suggest that NZ has over 1200 teleost fish species and high endemism (21.7%). Approximately 6% of NZ's coastal fishes have a tropical or subtropical affinity (Roberts, Stewart, & Struthers, 2015). However, many of these species are not resident or established (Francis & Evans, 1992) and could serve as indicators of biodiversity responses to climate change. Tropical or subtropical fishes have been recorded sporadically in NZ waters over the last 100 years and are anecdotally associated with nearshore Sea Surface Temperature (SST) anomalies which result from long-term variations in the El Niño Southern Oscillation (ENSO; Francis & Randall, 1993; Francis, Worthington, Saul, & Clements, 1999), variability in the strength and location of the EAUC (Laing, Stanton, & Challenor, 1996; Stanton & Sutton, 2003) and local weather patterns (Bowen et al., 2017; Sutton et al.,2005). Tropical and subtropical fishes may be among the first observable indicators of climate-induced change to the NZ's marine biodiversity; however, which species are undergoing poleward range shifts in NZ is poorly understood, let alone their potential impacts on resident communities and maritime industries.

Here, we examine the identities and occurrences of tropical, subtropical and rare fishes in NZ waters. First, we consolidate records from diverse sources to identify 'focal species' and describe new species records for NZ. We increase the spatial and temporal resolution of data for these focal species by including unpublished records and observations solicited from citizen scientists over the last 50 years. Second, we characterise the contribution of our focal species to the fish biodiversity of

NZ, providing a view of how the nation’s fish fauna may be changing. Third, we examine spatio-temporal trends in the distribution and number of focal species occurrences. We expect that the number of focal species occurrences, and the diversity of these focal species, has increased over recent decades, owing to climate change. Furthermore, we expect that the prevalence of these focal species within mainland NZ may vary, revealing regions or locations where overall diversity is disproportionately increased, or where certain families, or life-stages dominate the occurrences. Our study identifies taxa and geographic regions that are likely to undergo change in future and provides a baseline for future monitoring of NZ’s fish fauna.

2.3 *Methods*

2.3.1 Focal species

We focussed on marine ray-finned fishes (Class Actinopterygii) found in New Zealand (NZ) waters, excluding the sub-Antarctic islands and the Rangitāhua/Kermadec Islands groups (hereafter “mainland NZ”) (Figure 1). We derived a list of focal species by searching the two most comprehensive guides to NZ fishes (Francis, 2012; Roberts et al., 2015 and subsequent updates Roberts et al., 2017, 2020) for all species described using any of the following keywords: vagrant, rare, infrequent, occasional, tropical, subtropical, immigrant, and uncommon. We further refined this focal species list to fishes that could be observed or caught shallower than 50m depth according to Fishbase (www.fishbase.org), OBIS (https://obis.org/), or published literature. Migratory species that appear annually in mainland NZ albeit seasonally, were excluded (e.g., Striped marlin, *Kajikia audax*), but we included migratory species that vary in their year-to-year appearance (e.g., Wahoo, *Acanthocybium solandri*).

2.3.2 Focal species occurrence data

We collated occurrences of focal species from peer-reviewed literature, unpublished records held by scientists, underwater observations, photographic records and other unpublished anecdotal information. The majority of the focal species records originated from opportunistic citizen science observations, including those occurrences solicited using the WhatsthatfishNZ Facebook page (www.facebook.com/WhatsthatfishNZ) and sightings sourced from inaturalist NZ (https://inaturalist.nz/). Data sources varied in type, content, and completeness; hence, we developed a two-stage validation system to ensure consistency and reliability of the data accompanying an occurrence. First, we required that each occurrence was accompanied by some minimum metadata (including location with at least 10km precision, year and season of the occurrence along with the number of individuals sighted), and a photographic image, so that expert scientists or taxonomists could verify the identity of the focal species. We did not require an image if the occurrence data were sourced from experts directly, including specialist marine scientists or experienced members of the community (Figure 2). For occurrences that only included a location name, we used Google Earth (http://www.earth.google.com) to determine spatial coordinates, including a measure of coordinate uncertainty. Where possible, observers were asked to provide additional metadata including: the sex, life-stage, size estimate (or measurement from photo), water depth, depth of occurrence, as well as the habitat.

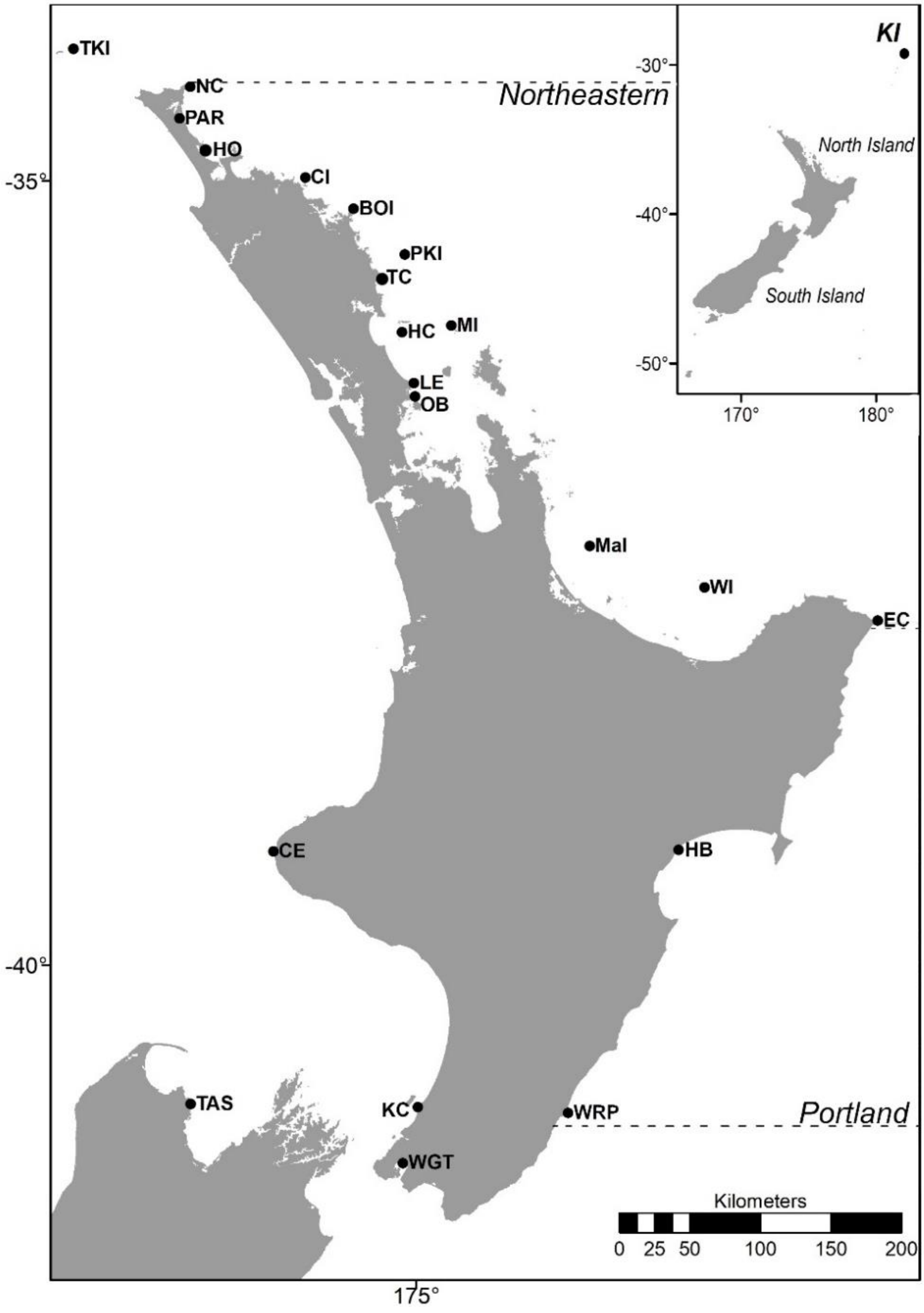


Figure 2.1: New Zealand, including North-eastern and Portland bioregions (indicated by the dotted lines) and locations mentioned in text, including: offshore island groups (CI: Cavalli Islands, HC: Taranga/Hen and Chicken Islands, MaI: Tuhua/Mayor Island, MI: Mokohinau Islands, PKI: Tawhiti Rahi/Poor Knights Islands, TKI: Manawatāwhi/Three Kings Islands, WI: Whakaari/White Island) and coastal locations (CE: Cape Egmont, EC: East Cape, HB: Hawkes Bay, HO: Houhora Harbour, BOI: Bay of Islands, KC: Kapiti Coast, LE: Leigh, NC: North Cape, OB: Omaha Bay, PAR: Parengarenga Harbour, TAS: Tasman, TC: Tutukaka Coast, WRP: Wairarapa, WGT: Wellington,). Inset: a potential subtropical and tropical source location for focal species (KI: Rangitāhua/Kermadec Islands).

2.3.3 New-to-New Zealand species

Any validated occurrences of fish species that had not previously been recorded in mainland NZ according to peer-reviewed publications (i.e., Roberts, Stewart, Struthers, Barker, and Kortet (2020); Roberts et al. (2015), and Francis (2012) were considered to be new-to-NZ species and were added to our list of focal species. We assigned the new-to-NZ species a biogeographic affinity based on their occurrences in OBIS (https://obis.org/) and categories proposed by Roberts et al. (2015): *Widespread:* occurs in the Pacific, Indian and Atlantic Oceans, or in the Pacific and Atlantic Ocean; *Endemic*: limited to NZ; *Australasian*: distribution mostly centred around Australia, New Caledonia, The Tasman Sea and NZ; *Indo-Pacific:* occurs in the Indian and Pacific Oceans; *Tropical and southwestern Pacific:* limited to the Pacific Ocean between 40^{o}N and 40^{o}S. If there were inconsistencies among data sources, we assumed the broadest distribution for each species. To visualise the contribution of these biogeographic regions to the new-to-NZ taxa, we plotted the kernel density of their cumulative occurrences using the stat_density_2d function in ggplot2 (Wickham, 2016) in R (v1.1-8), using Rstudio Version 1.1.456.

2.3.4 Contribution to New Zealand fish biodiversity

The contribution of focal species to NZ's marine teleost diversity was quantified at the species, and taxonomic family level. For each family that contained a focal taxon, we assessed the contribution of focal species to the total diversity in that family, relative to 'native NZ taxa' (i.e., species that occur in NZ but are not focal species for this study). We calculated and plotted species richness (i.e., the total number of NZ native and focal species in each family) and proportional contribution of focal and native taxa to diversity using the R base statistics package (R Core Team, 2020)

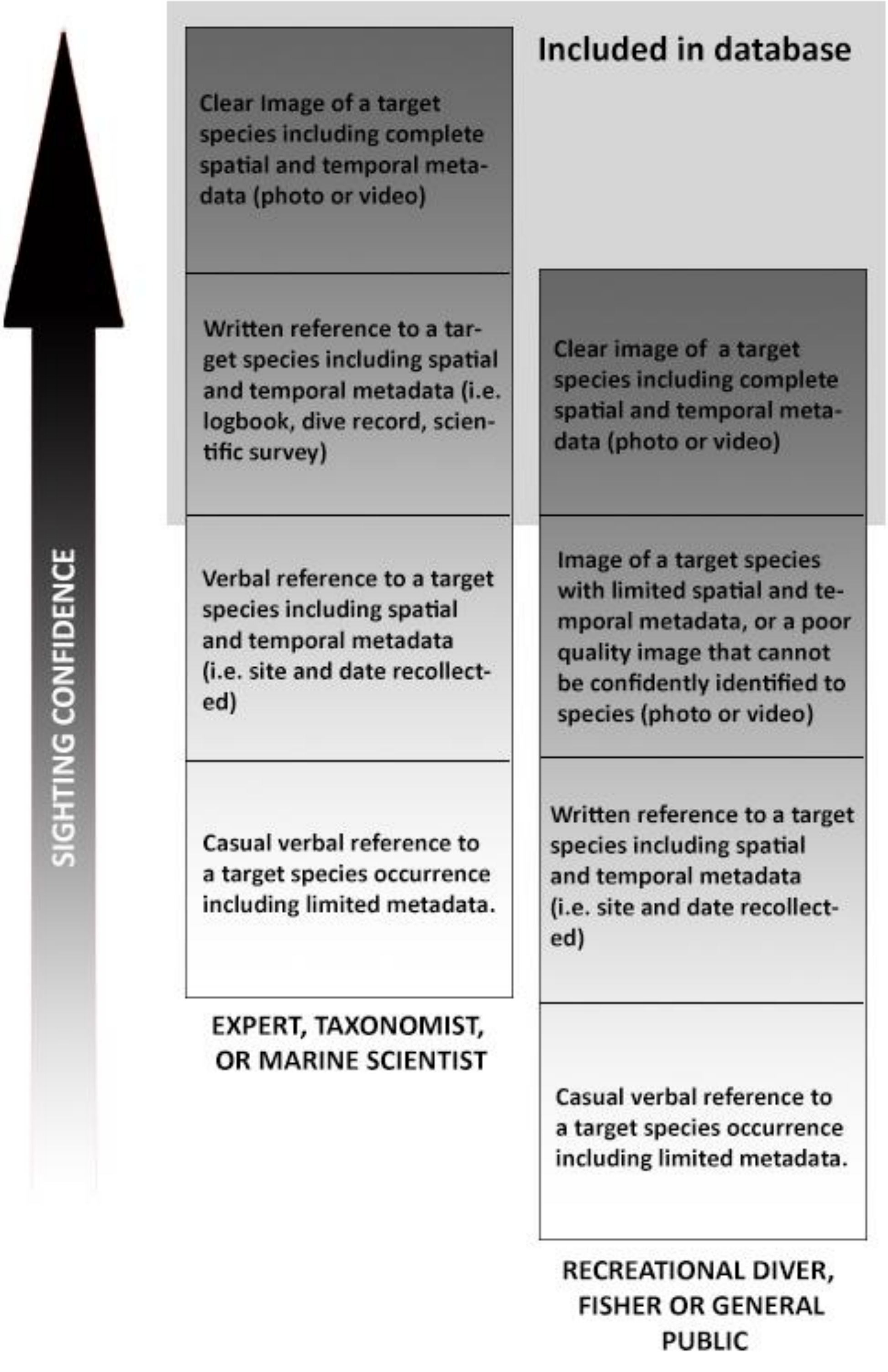


Figure 2.2 Method for validating and including species occurrences in the focal species database and subsequent analysis of spatio-temporal occurrences of tropical, subtropical, and rare fish species in temperate New Zealand. A lower level of confidence was required for occurrences sourced from expert scientists.

2.3.5 Spatio-temporal patterns within focal species records

To investigate spatio-temporal patterns in the number of occurrences and the species richness represented by those occurrences, we mapped all focal species records using ArcMap 10.7.1 (Esri Inc., Redlands, CA). We analysed richness and occurrences by decade (to 1970, 1971 to 1980, 1981 to 1990, 1991 to 2000, 2001-2010, 2011 to now), austral season (winter, summer, spring, autumn), life-history stage (juvenile or mature), habitat (benthic or pelagic), taxonomic family, and whether they were within a family with no native NZ species. For each subset of the focal species records, all occurrences within a 5 km tolerance were pooled to represent the same location and assigned a new common coordinate value using the 'Integrate' tool in ArcMap 10.7.1. We then created a weighted point for each location by combining the coincident occurrences (providing the total number of occurrences) or the number of unique species (providing focal species richness).

To identify areas that have a greater number of occurrences, or higher species richness, relative to other surrounding areas, we conducted 'hotspot' analyses (e.g., Betty et al. 2020; Aben et al. 2012; Isobe et al. 2015; Lepers et al. 2005) using the Getis-Ord Gi* statistic (Getis & Ord, 1992; Ord & Getis, 1995). As should be expected for citizen-sourced observations, our occurrence locations were not uniformly distributed across our seascape and hence the default number of nearest neighbours (K) used in hotspot analysis required consideration. We balanced the number of nearest neighbours that would allow us to estimate the Getis-Ord Gi* statistic precisely with the number of neighbours that would allow us to make inferences at a sensible spatial scale. To select our value of K for our analysis, we first visualised the relationship between the mean and variance of the Euclidean straight-line distances between our sampling locations at different values of K. Although there was a marked change in the rate of increase of the variance of the mean at a value of K=8 (Appendix 1, Figure S1.1), the mean spatial distance between locations for K=8 was over just 20km which we considered too large to make local inferences, particularly as many offshore islands have distinctly different fauna and settings but are within 20km of the mainland. Hence, a value of K=4, which resulted in a mean spatial distance of 12km, was selected as it reduced the effect of noise on the classification and was considered more appropriate given the geographic spread and sparsity of our data in some areas. We used the nearest neighbourhood contiguity scores to construct a spatial weights matrix to inform the spatial hotspot analysis.

To determine areas of greater than expected concentrations of high numbers of focal species occurrences in coastal NZ or high focal species richness, we calculated the Getis-Ord Gi* statistic (Getis and Ord 1992, 1995) for each location using the LocalG function in the spdep package in R (v1.1-8, Bivand & Wong, 2018). The Gstari* statistic returned a z-score (standard deviation) for each location, and we calculated a post-hoc p-value for each location. For statistically significant, positive z-scores, the larger the z-score is, the more intense the spatial clustering of occurrences, or the clustering of high species richness locations (i.e., hot spots). Hot spots (higher numbers of focal species occurrences or species richness than expected by chance) were identified as locations with an associated p-value of 0.1 or less. Due to the presence only nature of our dataset we exclude cold spot analysis (negative z-scores) and only report hot spots here.

2.4 *Results*

2.4.1 Focal species and new-to-New Zealand records

We identified 151 focal species (Appendix 1, Table S1.1) and our database comprised 1,657 records for 126 focal species from 51 families, including pelagic, reef-dwelling, soft sediment, estuarine and shallow-water species that have varied biogeographic affinities (Appendix 1, Table S1.1). The records spanned more than 50 years with an increase in occurrence data after 1980 and extended from the Manawatāwhi/Three Kings Islands (TKI) at the top of the North Island, to the Tasman region (TAS) at the top of the South Island (from 42 to 34°S; Figure 1). Most of the records were for focal species that are currently known to occur in NZ waters (Francis, 2012; Francis & Evans, 1992; Francis et al., 1999; Roberts, Stewart, Struthers, Barker, & Kortet, 2020; Roberts et al., 2015). However, 13 of the recorded species were new-to-NZ and four species had been recorded at Rangitāhua/Kermadec Islands previously (Francis, 2019; Roberts et al., 2020) but were new-to-NZ mainland (Appendix 2, Table S2.1 and Figure S2.1). All but two of the new-to-NZ fishes were single individuals, the exceptions were a pair of Horned blennies (*Parablennius intermedius*) that were sighted on a wharf piling in Parengarenga (PAR, Figure 1, Figure 3B), and two Izu's hogfish (*Bodianus izuensis*) at Serpent Rock at Tawhiti Rahi/Poor Knights Islands (PKI, Figure 1, Figure 3B).

Additionally, most of the fishes sighted were juveniles, except for the Painted grinner (*Trachinocephalus trachinus*), Blue-striped fang blenny (*Plagiotremus rhinorhynchos*), Lord Howe moray (*Gymnothorax annasona*), Variegated lizardfish (*Synodus variegatus*) and Moorish idol (*Zanclus cornutus*) which were all large enough to be mature (according to FishBase, 2019) (Figure 3B). Many of the new-to-NZ fishes were only sighted once. However, the Barred filefish (*Cantherhines dumerilii)* was recorded at TKI in 2016 and then again 285km away off the Tutukaka coast (TC) in 2018, and Izu's hogfish has been recorded on numerous occasions at Serpent Rock, PKI. The initial occurrence in 2017 was of a juvenile approximately 60mm total length (TL), and in 2019 a pair of terminal phase (~160mm TL) Izu's hogfish were photographed in the same location. The Lord Howe moray was first sighted in 2014 at PKI in two different locations, and in 2018 observers also photographed this species at Deepwater Cove in the Bay of Islands (BOI) some 55km away. Before the verified record in 2019, there were two unconfirmed reports of the Bluestripe fang blenny at PKI in 1972 and 2018 (pers comm Wade Doak and Sophie Roselt-Helmsalk). The congener, the Piano fang blenny (*Plagiotremus tapeinosoma*) is relatively common at PKI and could be confused for the Bluestripe fang blenny; hence we consider the verified record from Autumn 2019 as the first confirmed occurrence of the Bluestripe fang blenny in NZ. Similarly, a previous report of a Painted grinner exists from Mangawhai, a coastal lagoon south of Whangarei in north-eastern NZ, however, this occurrence has not been published (C. Struthers., personal communication, 12 February 2020).

The majority of the new-to-NZ and new-to-NZ mainland species had Indo-Pacific (Giant frogfish, *Antennarius commerson*; Two-spot hogfish, *Bodianus bimaculatus*; Günther's butterflyfish, *Chaetodon guentheri;* Silver-cheeked toadfish, *Lagocephalus sceleratus*; Two-spot wrasse, *Oxycheilinus bimaculatus*, Blue-stripe fang blenny and Two-stripe goby, *Valenciennea helsdeingenii* and Moorish idol, *Zanclus cornutus*), or Tropical and southwestern Pacific affinities (Izu's hogfish*;* Pacific red-stripe hogfish, *Bodianus masudai;* Barred filefish; Barred knifejaw, *Oplegnathus fasciatus;* Horned blenny, *Parablennius intermedius*) (Figure 3, Appendix 2, Figure S2.2). The Lord Howe moray and the Painted grinner had Australasian distributions and the Tailor (*Pomatomus saltatrix*), and Variegated lizardfish were the only 'widespread' new-to-NZ and new-to-NZ mainland species (Figure 3B).

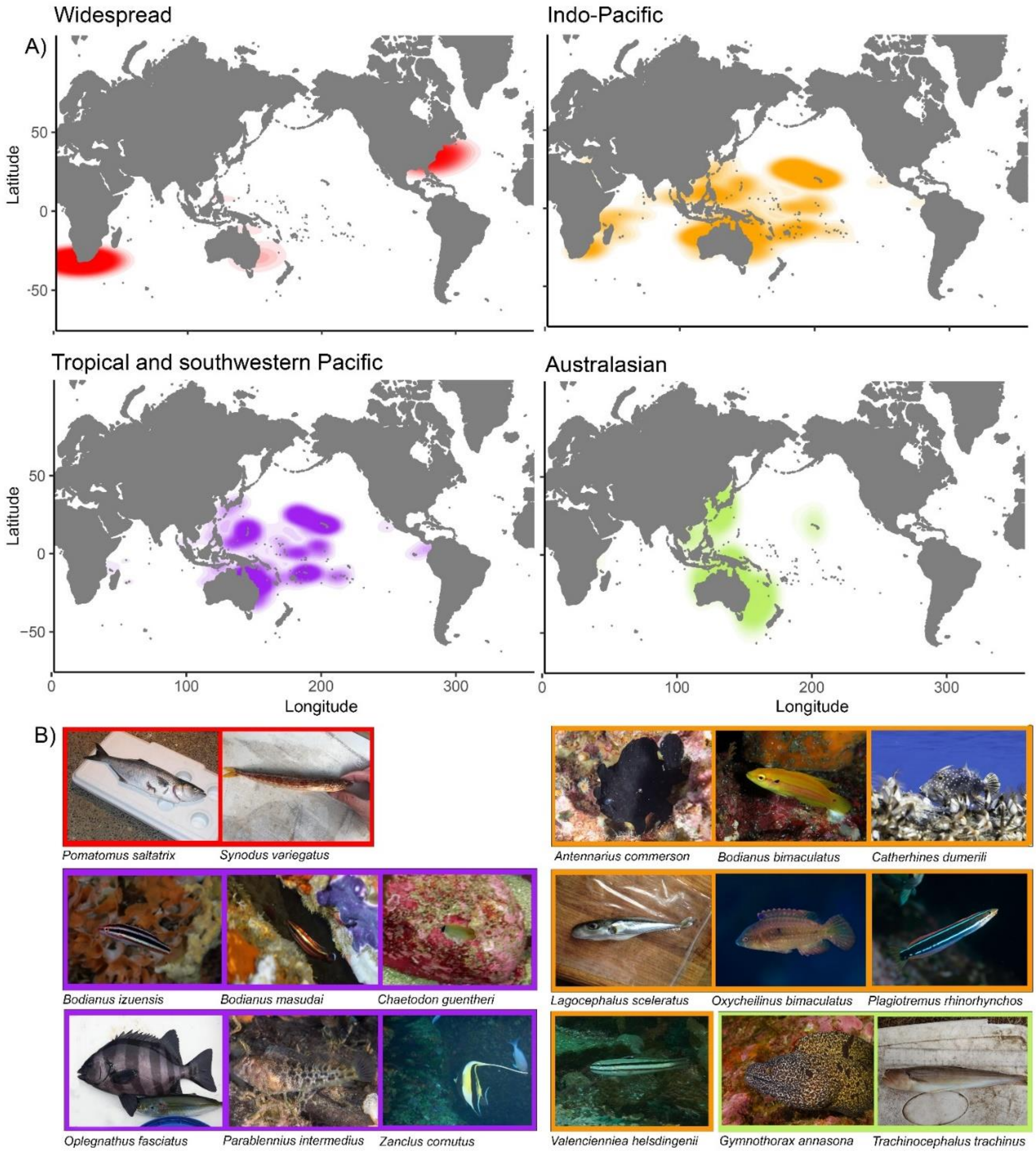


Figure 2.3 A) Joint distribution of new-to-New Zealand (NZ) and new-to-NZ mainland species collated by biogeographic affinity, darker areas have a higher number of occurrences of the relevant focal species (occurrence data sourced from OBIS; 2021). B) Original images used to validate new-to-NZ species. Coloured borders denote the biogeographic affinity of each species.

2.4.2 Contribution of focal species and new-to-New Zealand species to fish biodiversity

The focal species for which we have occurrence data represent 25 percent of NZ’s Actinopterygii families, and focal species account for 6.5% of all known pelagic species and 17.3% of all known benthic species occurring in less than 50m depth in NZ’s oceans. Of the 50 families that included focal species, 25 families were represented only by focal species and 26 also contained NZ native species, although 16 of these ‘mixed’ families were represented by two or less native species (Figure 4). Most of the families that included high proportions of focal species were from the Order Perciformes (Appendix 1, Table S1.1), with the highest focal species richness occurring in the Labridae (*n* = 21) and Pomacentridae families (*n* = 11). Of the families that had both NZ native and focal species present, those with the highest proportion of focal species were the Monocanthidae (85.7%), Pomacentridae (84.6%) and Blenniidae (80%) (Figure 4). The focal species varied in habitat preferences; seven of the 50 families with focal species had records of focal species that were exclusively pelagic, while 36 families with focal species contained records of benthic focal species (Appendix 1, Table S1.1).

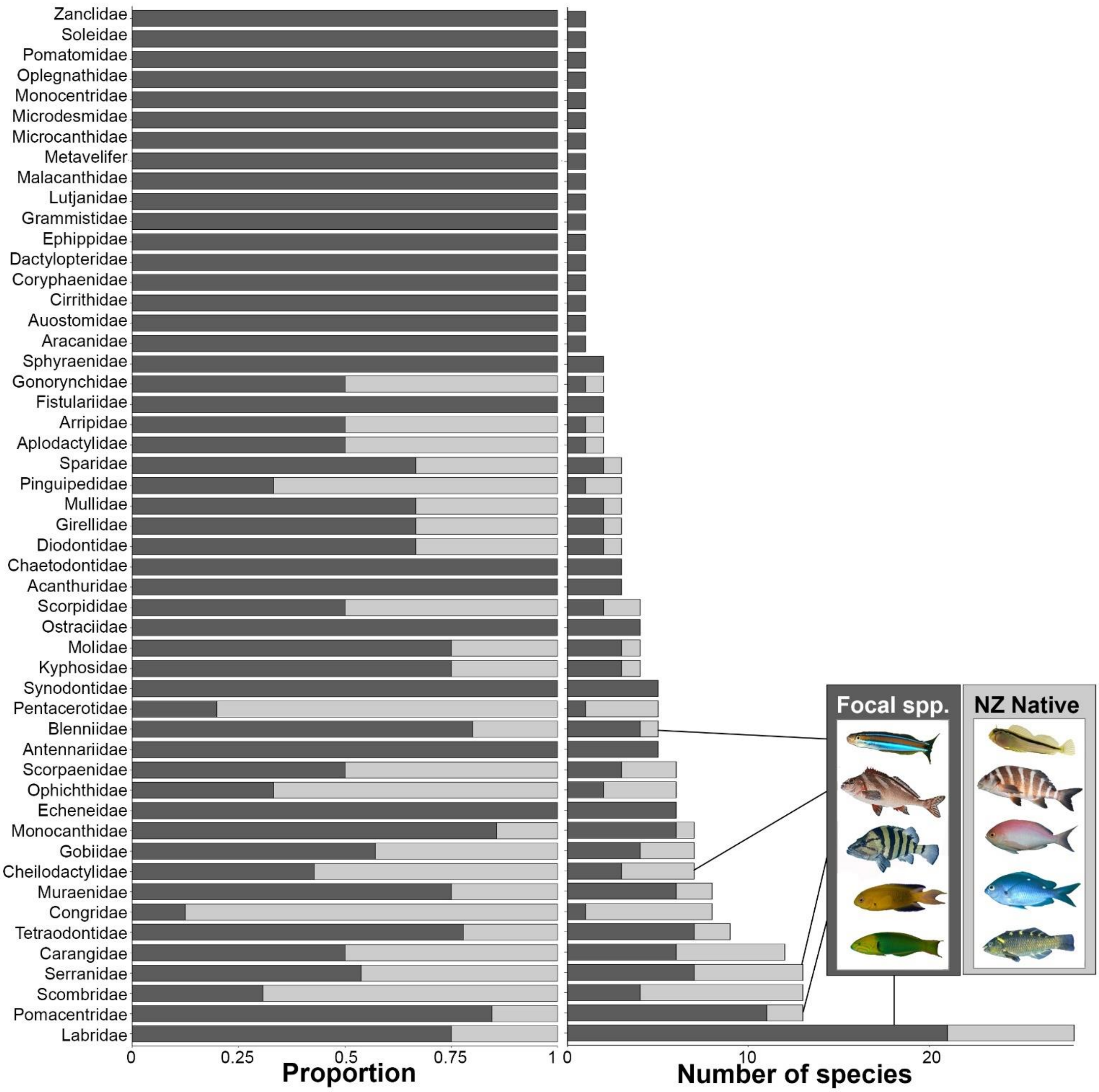


Figure 2.4 The proportion and number of focal species (i.e., new-to-New Zealand (NZ), tropical, subtropical and rare species, black shading) and NZ con-familial native species (NZ native, i.e., species found in NZ that are in the same taxonomic family, grey shading) for each family that has at least one focal species. Example images of focal species and NZ native taxa are shown for families discussed in the main text.

2.4.3 Spatio-temporal patterns in occurrences and diversity

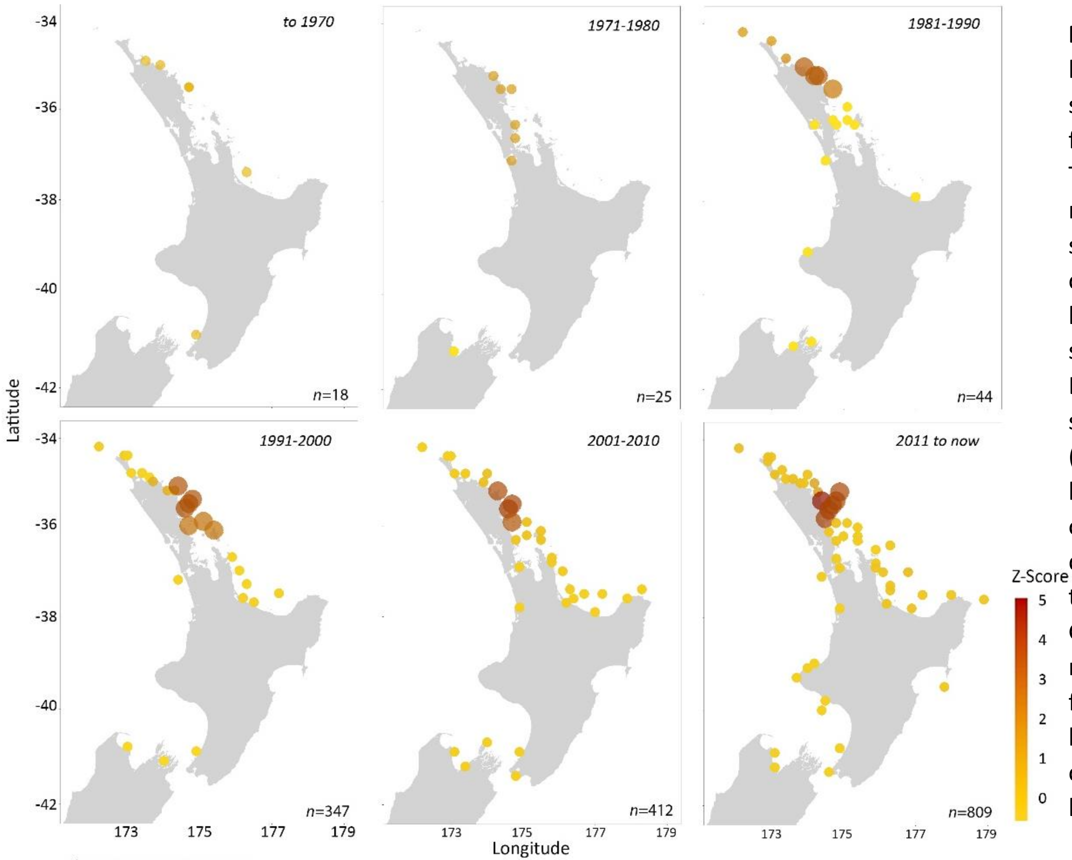
Focal species occurrences and diversity has increased over the past five decades (Figure 5). Before 1980 there was no apparent spatial clustering of records for the focal species (Figure 5A) or species richness (Appendix 3, Figure S3.1) likely due to the paucity of historical data. After 1980, occurrence and species richness hotspots become more consistent, with hotspots clustered in north-eastern NZ. A persistent hotspot of occurrences is identified in the region of PKI that becomes more prominent after 1990 (1980 to 1990, Z = 1.574, p = 0.06; 1991 to 2000, Z=2.613, p=<0.01; 2001 to 2010, Z=2.922, p=<0.01; 2011 to now, Z=2.871, p=<0.01), however a hotspot in species richness at PKI was only identified from 1991 onwards (1991 to 2000, Z=2.669, p=<0.01; 2001 to 2010, Z=3.343, p=<0.01; 2011 to now, Z=2.303, p=0.01; all spatial hotspots of richness are presented in Appendix 3). Additionally, BOI was a statistically significant spatial hotspot of occurrences between 1980 and 2001 (Z=2.329, p=<0.01; Z=2.553, p=<0.01). Species richness hotspots were identified in the BOI between 2001 to 2010 (Z=3.524, p=<0.01) and from 2011 to now (Z=1.540, p=0.06). More southerly hotspots of focal species occurrences were located at Aotea/Great Barrier Island (GBI; Z=1.743, p=0.04), Mokohinau Islands (MI; Z=1.802, p=0.04) and Taranga/Hen and Chickens Islands (HCI; Z=1.802, p=0.04) from 1991 to 2000, however only the HCI hotspot persisted from 2001 to 2010 (Z=2.588, p=<0.01) (Figure 5). Between 1981 and 1990 species richness hotspots were identified in the Bay of Plenty (BOP; Z=2.512, p=<0.01) and GBI (Z=1.676, p=0.05).

When analysed independently, the occurrences of both mature and juvenile fishes were consistently clustered around the north-eastern coast of NZ, specifically the TC (mature: Z=2.571, p=<0.01, juvenile: Z=1.992, p=<0.01) and PKI (mature: Z=2.571, p=<0.01, juvenile: Z=1992, p=<0.01) (Figure 6). However, mature species occurrences extended further south than those of juvenile fishes on both the east and west coast of the North Island, with mature fish occurrences located on the Kapiti Coast (KC), Wellington (WGT) and in Hawke Bay (HB; Figure 6). Juvenile species richness followed a similar pattern to occurrences, with hotspots centred around PKI (Z=2.115, p=0.02) and TC (Z=2.115, p=0.02). However, mature focal species richness was more widespread with hotspots at

PKI (Z=1.410, p=<0.08) and further north in BOI (Z=1.893, p=0.03) and Cavalli Islands (CI, Z=2.054, p=0.02, Appendix 3, Figure S3.2).

Pelagic species and benthic species occurrences and species richness varied spatially. Pelagic species occurrences were spread further south along the east coast of the North Island than benthic species occurrences, and benthic species occurred further south along the west coast (Figure 6). Spatial hotspots for pelagic species occurrences and richness were located further south than those of benthic species, at HCI (occurrences: Z=2.186, p= 0.01, richness: Z=2.909, p=<0.01), whereas occurrence and richness hotspots for benthic species varied. Occurrences were centred around the TC (Z=2.874, p=<0.01) and PKI (Z=2.874, p=<0.01), and species richness hotspots were centred further north at BOI (Z=2.305, p=0.01) (Appendix 3, Figure S3.2).

A) OCCURRENCES



B) SPECIES RICHNESS

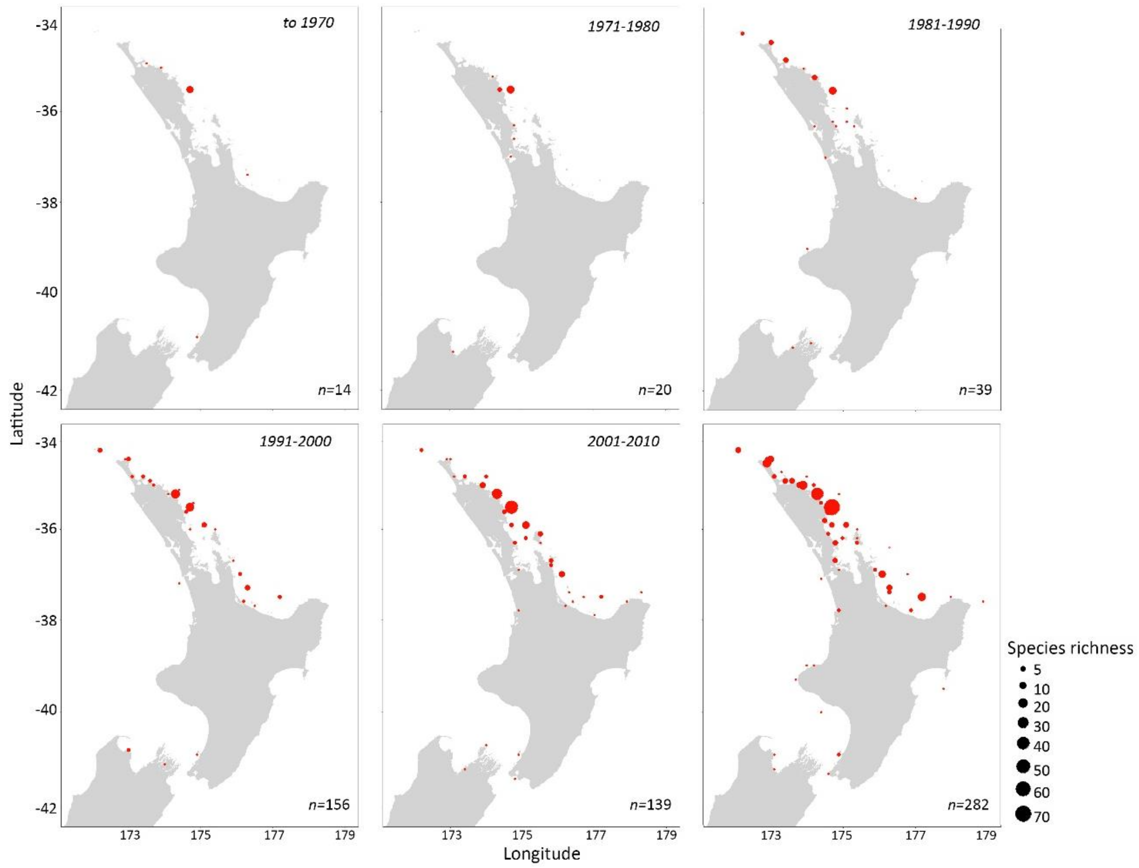


Figure 2.5A) Spatial hotspots in focal species occurrences from 1958 to 2020. The colour gradient represents the z-score, where darker colours denote locations of high species occurrences. Larger points are statistically significant (p ≤ 0.10) clusters of high species occurrences determined using are the Getis-Ord Gi*statistic. B) The number of individual focal species per location, per decade clustered by five-kilometre radius

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Total number of focal species occurrences peaked during the austral summer (*n*=389) and autumn (*n*=824) and was lowest during the spring (*n*=201) (Figure 6). However, seasonal changes in the spatial distribution of focal species occurrences and richness were minimal; PKI (occurrences, Figure 6: Summer Z=3.869, p=<0.01; Autumn Z=2.579, p=0.01; Spring Z=2.121, p=0.03; Winter Z=2.177, p=0.03; richness, Appendix 3, Figure S3.3: Summer Z=3.568, p=<0.01; Autumn Z=1.456, p=0.07; Spring Z=2.459, p=<0.01; Winter Z=2.237, p=0.01) and nearby TC (occurrences, Figure 6: Summer Z=2.359, p=0.02; Autumn Z=2.585, p=0.01; Spring Z=2.121, p=0.03; Winter Z=2.142, p=0.03, richness, Appendix 3, Figure S3.3: Summer Z=2.156, p=0.02; Autumn Z=1.456, p=0.07; Spring=2.459, p=<0.01; Winter Z=2.171, p=0.01) were consistent hotspots throughout the seasons. Additionally, during the austral spring, spatial clustering of occurrences (Z=2.159, p=0.03) and species richness (Z=2.191, p=<0.01) were observed at BOI, and occurrences were clustered further south at HCI (occurrences: Z=1.941, p=0.03) (Figure 6). During the winter both occurrence and species richness hotspots were also observed at HCI (occurrences: Z=1.709, p=0.04; richness: Z=1.812, p=0.03) (Figure 6, Appendix 3, Figure S3.3).

Families with predominantly tropical affinities (Labridae and Pomacentridae) were restricted to north-eastern NZ with only a single occurrence of pomacentrids in the South Island (Figure 6). There were several spatial hotspots of occurrences for both pomacentrids and labrids which included BOI (Pomacentridae Z=1.950, p=0.05; Labridae Z= 1.949, p=0.03), PKI (Pomacentridae Z=1.911, p=0.06; Labridae Z=1.994, p=0.02) and the CI (Pomacentridae Z=1.950, p=0.05; Labridae Z=1.949, p=0.03).

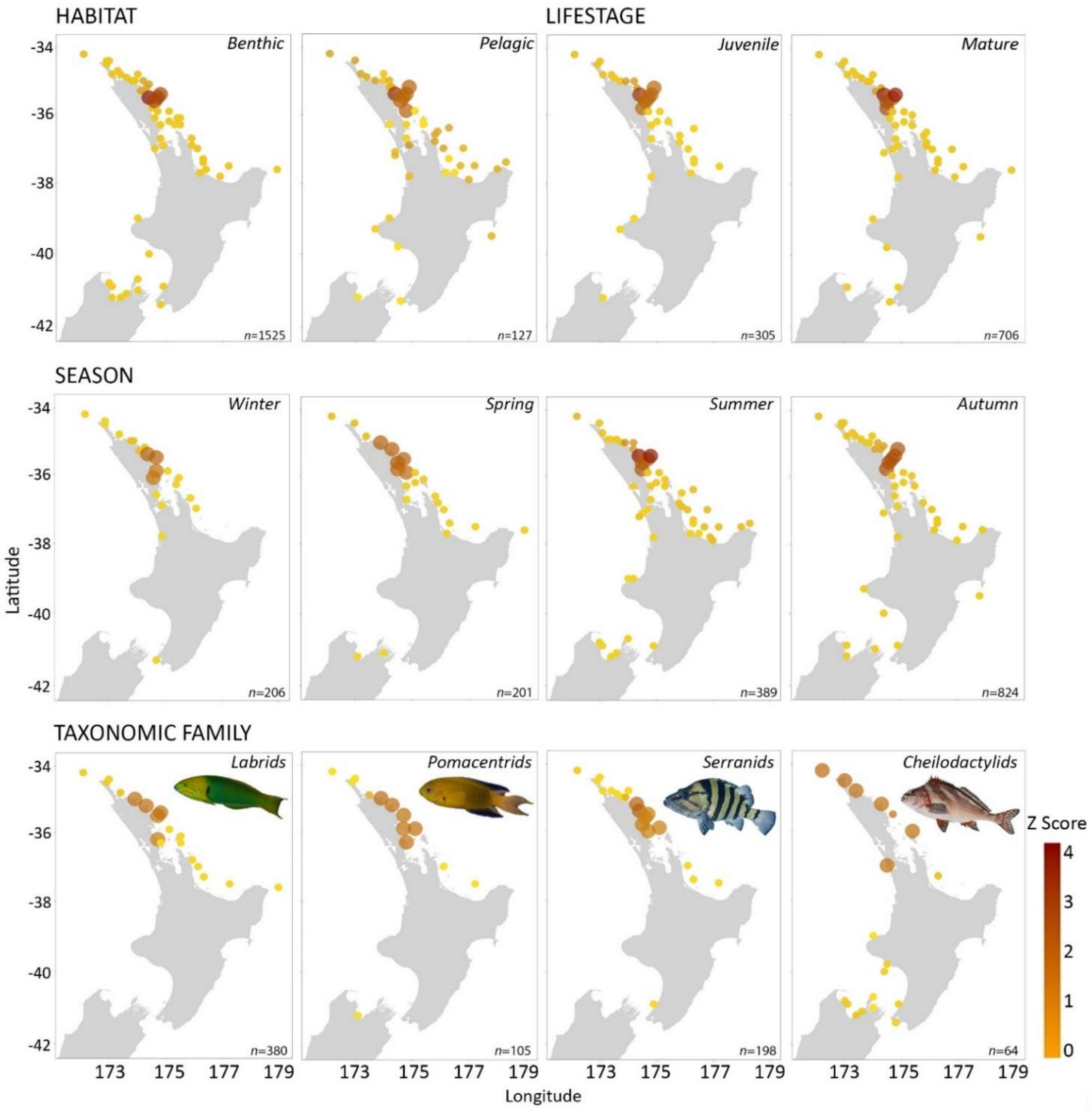


Figure 2.6: Spatial hotspots of focal species occurrences grouped by Season (Austal Summer: December to February; Autumn: March to May; Winter: June to August; Spring: September to November), Habitat (Pelagic and Benthic) and some of the studied taxonomic Families (including a photo of an example focal taxon). The colour gradient represents the z-score, where darker colours denote locations of high species occurrences. Larger points are statistically significant ($p \leq 0.10$) clusters of high species occurrences determined using are the Getis-Ord Gi*statistic.

Pomacentrids were also significantly clustered further south at HCI (Z=1.808, p=0.03), MI (Z=1.808, p=0.03) and Leigh (LE; Z=1.808, p=0.03) (Figure 6). Occurrence hotspots were the most widespread for the Cheliodactylidae, from the TKI (Z=2.022, p=0.02) in the north and GBI (Z=1.778, p=0.04) in the south. Cheilodactylidae was also the only family with an occurrence hotspot on the west coast of NZ, at the Manukau Heads (Z=1.778, p=0.04, Figure 6).

2.5 *Discussion*

The occurrence of tropical fishes in temperate regions is recognised as one of the first indicators of climate-mediated biodiversity change (Munday, Jones, Pratchett, & Williams, 2008; Perry, Low, Ellis, & Reynolds, 2005). In contrast to surveying species loss and abundance decline, recording the presence of novel and highly observable species – such as tropical fishes – provides us with a means to document biodiversity change in a way that can easily engage the public. Researchers have observed tropical and subtropical fishes in New Zealand (NZ) waters for over 100 years (Francis, 1988; Francis & Evans, 1992; Francis et al., 1999; Moreland, 1959), however, few studies have consolidated their contribution to NZ’s marine diversity. Here, we present an up-to-date analysis of occurrence data for tropical, subtropical and rare fishes from previously unpublished and novel data sources, including citizen science observations. We identify 151 tropical, subtropical and rare teleost fish species (focal species) in NZ waters and present 14 new-to-NZ species and 3 new-to-NZ mainland species previously recorded at Rangitāhua/Kermadec Islands. Our study reveals the potential for these species to be monitored as indicators of climate change impacts and highlights the utility of citizen-sourced occurrence data to provide baseline information for these indicator species and as a means for monitoring their occurrences. Furthermore, we identify specific locations where the occurrences and the richness of focal species were concentrated, providing suitable locations for ongoing monitoring of indicator species and their potential impacts.

Globally, temperate reefs at similar latitudes to NZ have undergone significant biodiversity shifts due to climate change (Poloczanska et al., 2016; Ramírez, Afán, Davis, & Chiaradia, 2017; Sunday et al., 2015; Vergés et al., 2014; Wernberg et al., 2016). For example, an increase in North Sea fish diversity was driven by poleward range shifts of warmer water fishes occurring more rapidly than the rate of local cold-water species losses (Hiddink & ter Hofstede, 2008). In Tasmania, the increased abundance of 52 coastal fish species (Last et al., 2011) and the establishment of previously rare or absent species have been attributed to climate change (Stuart-smith, Barrett, Stevenson, & Edgar, 2010). Our results confirm that 25.4% of NZ's teleost fish diversity are tropical, subtropical, and rare fishes (i.e., our focal species) and that the contribution of these fishes has increased over the last 50 years. Furthermore, we identified 25 families with tropical, subtropical or rare fishes that contained no NZ native species (Figure 4), representing distinct taxonomic contributions to NZ's teleost fish diversity. The new-to-NZ species identified in our study increased the contribution of the Tropical and southwest Pacific (+0.36%) and Indo-Pacific (+0.4%) regions relative to the last complete assessment of the NZ's fish fauna in 2013 (Roberts et al., 2015). The increase in the contribution of warmer water taxa to NZ's fish diversity suggests that a climate-mediated shift in NZ's fish biodiversity is underway, eroding the distinctiveness of this temperate biogeographic region and signalling potential future broadscale ecological changes (*sensu* Fogarty, Burrows, Pecl, Robinson, & Poloczanska, 2017).

The redistribution of pelagic species globally has been more rapid than benthic species (Champion, Brodie, & Coleman, 2021). For example, in eastern Australia, the rate of poleward range shifts of pelagic species like Kingfish (*Seriola lalandi*, 94.4km per decade (Champion, Hobday, Tracey, & Pecl, 2018) and Mahimahi (*Coryphaena hippurus*, 148.7km per decade (Champion et al., 2021) was three to five times faster than the average rate of range shifts for benthic species in the same region (38km per decade; Sunday et al. 2015). Pelagic species typically have dynamic range edges, where distributional changes respond rapidly to changing environmental conditions (Sunday et al., 2015); as a result, species-specific adult mobility has been recognised as a significant predictor for the rate of range extensions (Feary et al., 2013; Champion, Brodie, & Coleman, 2021). We found that pelagic focal species in NZ waters were more broadly distributed than benthic focal species (Figure 6). Hotspots of pelagic occurrences occurred 80km further south than benthic species occurrence hotspots, and several pelagic occurrences were located south of East Cape, a known biogeographic boundary (Shears et al. 2008; Figure 6). In contrast to pelagic species, benthic species are less mobile, and their capacity to undergo climate-mediated range shifts depend on their species-specific ability to disperse as larvae, find appropriate settlement habitat, and tolerate local conditions to survive (Bates et al., 2014). In isolated oceanic island nations like NZ that are distant from sources locations, extralimital occurrences or range shifts of pelagic species are the most likely early warning signs of ocean climate change impacts.

Proportionally, the Labridae (Wrasses), Tetradontidae (Pufferfishes), and Pomacentridae (Damselfishes) families contributed the most tropical and subtropical species to NZ’s fish biodiversity (Figure 4). These results align with occurrences of tropical and subtropical fishes at similar latitudes; in temperate regions, the Pomacentridae and Labridae are recognised as disproportionally contributing to poleward range shifts by dispersing to and establishing viable populations (Feary et al., 2013). Labrids and pomacentrids are highly speciose and represent a broad range of functional groups, dietary and habitat preferences (Parenti & Randall, 2018; Stuart-Smith, Mellin, Bates, & Edgar, 2021; Westneat & Alfaro, 2005). Furthermore, geographic distributions of labrid and pomacentrid species are highly responsive to ocean temperature changes and are known to be early responders to climate change and have high dispersal potential (Feary et al., 2013; Cure et al., 2018; Shalders, Saunders, Bennett, Parker, & Harvey, 2018; Parker et al., 2019). Labrids, in particular, have relatively long pelagic larval durations (Victor, 1986), and several pomacentrid species are known to raft with marine debris, increasing their dispersal potential (Jokiel, 1990; Luiz, Allen, Robertson, Floeter, & Madin, 2015; Thiel &

Haye, 2006). Although evidence suggests that many wrasse and damselfish species are cold-tolerant and can over-winter and persist in temperate environments like NZ (Booth et al., 2018), we found that they are geographically restricted in NZ waters. Occurrences of families with tropical and subtropical affinities such as labrids and pomacentrids were spread along the Northland coast from Whangaroa south to the Hauraki Gulf (Figure 6). However, families with more temperate affinities like the Cheilodactylidae were more widespread throughout the North Island and down to the top of the South Island (Figure 6). The disparity in the ranges of families with tropical and temperate affinities suggests that NZ's thermal regime may be restricting the rate and latitudinal extent of poleward range shifts by tropical and subtropical fishes in our waters.

Most tropical, subtropical, and rare fish occurrences were centred around north-eastern NZ in the Northland and Portland bioregions (Shears, Smith, Babcock, Duffy, & Villouta, 2008; Figure 5). These bioregions are recognised as some of the most speciose in NZ and coincide with previous observations of rare and vagrant fishes in NZ waters (Francis, 1996; Francis et al., 1999; Roberts et al., 2015). North-eastern NZ is under the influence of the warm, poleward East Auckland Current (EAUC), an oceanographic feature often implicated in the dispersal of species to NZ from recognised source populations including Norfolk Island, Lord Howe Island and mainland Australia (Francis et al., 1999). Furthermore, the coastal waters off north-eastern NZ are consistently the warmest in NZ, suggesting that they are the most like the temperatures experienced by tropical and subtropical fishes in source locations. The sea surface temperature in north-eastern NZ peaks at 23.8°C (at Leigh) and the annual average is ~17°C (at Leigh, Ministry for the Environment & StatsNZ, 2015), which is the recognised thermal minima for survival of tropical reef fishes (Booth, Beretta, Brown, & Figueira, 2018; Figueira & Booth, 2010).

In temperate regions, many extralimital tropical and subtropical fishes undergo significant population bottlenecks during temperate winters and do not persist past juvenile life-stages

(Feary et al., 2013). Therefore, the presence of mature individuals indicates broad thermal tolerances and a capacity to overwinter and potentially breed in novel locations (Feary et al., 2013; Middleton et al., 2020). We found subtle differences in the distribution of mature and juvenile fishes. In general, the diversity and occurrence of mature fishes were higher than juvenile occurrences, and occurrences of mature individuals were further south than juvenile fishes (Figure 5). These results may seem counterintuitive given the thermal limitations of tropical and subtropical fishes in temperate regions. However, the occurrence hotspots of mature fish occurrences and the broad spatial distribution of mature fishes can be explained by the numerical dominance of subtropical Australasian families that have broader thermal tolerances (e.g., Arripidae, Cheilodactylidae and Microcanthidae) than the more tropical Indo-Pacific or Tropical and southwestern Pacific families (e.g., Aulostomidae and Cirrithidae; Supplement 1, Table S1.2).

When populations are limited by recruitment from distant tropical and subtropical source populations, the long-term persistence of large or mature fishes in temperate locations can provide an extinction buffer between stochastic recruitment or dispersal events. For example, mature Lord Howe coralfish (*Amphichaetodon howensis*) have been recorded in north-eastern NZ since the 1970s (e.g., Russell, 1971; Francis, 1996). However, occurrences of juveniles are infrequent (Francis, 2012), and breeding behaviour has never been observed, suggesting that the population primarily consists of mature non-breeding individuals with sporadic recruitment from tropical and subtropical source populations occurring rarely. Our results suggest that local thermal regimes do not constrain long-distance dispersal by tropical and subtropical fishes to NZ. However, the ability of warmer water species to persist or reach maturity could be limited by local temperature. As long-term warming increases in NZ, there will likely be an increase in the rate of overwintering and persistence of immigrating tropical and subtropical species, and the establishment of breeding populations for those species already persisting.

In our study area, native labrid and pomacentrid distributions overlap with the hotspots of focal congeners (Brook, 2002; Francis, 1996). We can assume that when native and novel species have similar resource requirements, competition will impact at least one of the two interacting species' survival (Sorte et al., 2010). Since most endemic fishes are specialists regarding habitat, environmental variables, and diet, these species may have a reduced ability to adapt to habitat changes, increased competition and predation by tropical and subtropical range shifting species (Ben Rais Lasram et al., 2010; Manes et al., 2021). Novel species interactions could lead to losses of local specialist species in favour of range-shifting generalists (Monaco et al., 2020). Although a greater number of generalist species could increase the ecosystem resilience to future disturbances, it is likely that essential and highly specialist ecosystem functions could be lost (Munday, 2004).

Long-term biodiversity monitoring is costly, time-consuming, and often spatially restricted (Hortal et al., 2015; Middleton et al., 2021). Citizen science – the participation of citizens in the collection of data or contribution of knowledge for the planning, development and execution of scientific research (Eitzel et al., 2017) – has been useful to overcome a lack of baseline data and detect range shifts and out-of-range occurrences in marine settings (Pecl et al. (2019); e.g., Range Extension Database and Mapping Project [Redmap] Reef Environmental Education Foundation [REEF]; European Alien Species Information Network [EASIN]). Our study increased the temporal and spatial resolution of available occurrence data by including data from diverse sources such as citizen science observations, previously unpublished data, and scientific observations. However, the inferences gained by our research may be, in part, determined by the spatial and temporal patterns of citizen science observations (Bird et al., 2014) and species conspicuousness (Middleton et al., 2021). Our results indicate that the number of observations has increased over time (Figure 6A), and hotspots of richness and occurrences may be hotspots of observational effort rather than spatio-temporal phenomena of biological significance. For example, our results identified the Poor Knights Islands, a marine protected area popular with recreational divers, as the most persistent and significant hotspot of focal species occurrences (Figure 6). Furthermore, small, and cryptic species in the Bleniidae, Gobiidae, and Microdesmidae only represented 4.7% of our observations, whereas large, conspicuousness and charismatic Labridae and Serranidae species represented 34.7% of the observations, suggesting observer bias toward larger, conspicuous species. Nonetheless, our data have provided an increased understanding of NZ's fish diversity and have identified fishes that may be indicators of climate change impacts on NZ's marine biodiversity.

Our data and findings have provided a baseline for future biodiversity change and identified locations where focal species are occurring, but further analysis and continued monitoring will clarify the specific species and physical and environmental drivers of long-term biodiversity change in NZ waters. Our results demonstrate that citizen science can increase our understanding of biodiversity. However, repeated sightings of individuals or species and observations of maturity or breeding are required to determine current species ranges and robustly determine range shifts (Middleton et al., 2021). Future research should focus on methods that encourage and motivate the long-term and repeated monitoring of specific focal species, life-stages and behaviours in areas of suspected biodiversity change.

Our results suggest that NZ's fish biodiversity has changed over the last half century, and it is likely to continue changing. Evidence suggests that the Eastern Australian Current's strength and southern penetration has increased in the last 30 years, causing a weakening of the Tasman Front (Cetina-Heredia, Roughan, van Sebille, & Coleman, 2014; van Gennip et al., 2017) that feeds the EAUC. These changes may cause a decrease in the immigration of tropical and subtropical fishes and invertebrates to NZ from Australia (Shears & Bowen, 2017). Instead, increased persistence of tropical and subtropical fishes may be facilitated by long-term solar warming of NZ waters and an increase in the frequency of marine heatwaves (Behrens, Fernandez, & Sutton, 2019; Law et al., 2018; Salinger et al., 2019). Our study provides a baseline for tropical, subtropical and rare species occurrences to date; these species are expected to be early indicators of climate-driven biodiversity change. Based on the baselines described here in, and as the observer effort of citizen scientists increases in future, NZ will be well-placed to infer climate-driven biodiversity change and to potentially mitigate escalating socio-economic impacts.

2.6 *Authors' Contributions*

IM, LL and JDA conceived and designed the study. IM, developed a focal species database, collected data from citizen scientists and developed data quality controls. LL and IM ran the Whatsthatfish Facebook page to interact with citizen scientists and collect species occurrence data. IM and JDA analysed the data and IM produced the figures. TT, MF, CS and CD contributed focal species occurrence data and assisted in the verification of record and species identifications. IM drafted the initial manuscript and all authors contributed towards the interpretations of results, the initial drafting and the revisions of the final manuscript.

2.7 *Appendix*

2.7.1 Supplement 1: Full focal species list including native confamilials and number of occurrences by biogeographic affinity.

Table S1.1: The full list of focal species for mainland New Zealand along with the native confamilials. Species shaded in grey are focal species for which we have records for in our database, those focal species for which we have no records and native species are not shaded. Biogeographic affinity and habitat preference are only provided for Focal species. Habitat preference was sourced from Fishbase (Froese, R., and D. Pauly. Editors. 2021.FishBase. World Wide Web electronic publication.www.fishbase.org, (02/2021)). "Benthic" includes those species found on soft sediment and reef substrates; "Pelagic" are those found in either nearshore or offshore pelagic habitats and "Benthic/pelagic" are those species that utilise both habitats continuously or during post settlement juvenile and mature phases. Biogeographic affinity was allocated based on definitions from Roberts et al., 2015 (Roberts, C., Stewart, A. L., & Struthers, C. D. (2015). The Fishes of New Zealand. Te Papa Press, Wellington NZ).

Order	Family	Focal species/NZ native	Species	Biogeographic affinity	Habitat preference
Perciformes	Acanthuridae	Focal species	*Acanthurus dussumieri*	Indo-Pacific	Benthic
Perciformes	Acanthuridae	Focal species	*Naso unicornis*	Indo-Pacific	Benthic
Perciformes	Acanthuridae	Focal species	*Prionurus maculatus*	Australasian	Benthic
Lophiiformes	Antennariidae	Focal species	*Antennarius commerson*	Indo-Pacific	Benthic
Lophiiformes	Antennariidae	Focal species	*Antennarius striatus*	Widespread	Benthic
Lophiiformes	Antennariidae	Focal species	*Antennatus nummifer*	Widespread	Benthic
Lophiiformes	Antennariidae	Focal species	*Fowlerichthys scriptissimus*	Indo-Pacific	Benthic
Lophiiformes	Antennariidae	Focal species	*Histrio Histrio*	Widespread	Benthic
Perciformes	Aplodactylidae	Focal species	*Aplodactylus etheridgii*	Widespread	Benthic
Perciformes	Aplodactylidae	NZ native	*Aplodactylus artidens*		
Tetraodontiformes	Aracanidae	Focal species	*Kentrocapros sp*	Unknown	Benthic
Perciformes	Arripidae	Focal species	*Arripis xylabion*	Australasian	Pelagic
Perciformes	Arripidae	NZ native	*Arripis trutta*		
Sygnathiformes	Aulostomidae	Focal species	*Aulostoma chinensis*	Indo-Pacific	Benthic
Perciformes	Blenniidae	Focal species	*Omobranchus anolius*	Australasian	Benthic
Perciformes	Blenniidae	Focal species	*Parablennius intermedius*	Tropical SW Pacific	Benthic
Perciformes	Blenniidae	Focal species	*Plagiotremus rhinorhynchos*	Indo-Pacific	Benthic

Perciformes	Blenniidae	Focal species	*Plagiotremus tapeinosoma*	Indo-Pacific	Benthic
Perciformes	Blenniidae	NZ native	*Parablennius laticlavius*		
Perciformes	Carangidae	Focal species	*Decapterus muroadsi*	Indo-Pacific	Pelagic
Perciformes	Carangidae	Focal species	*Elagatis bipinnulata*		Pelagic
Perciformes	Carangidae	Focal species	*Naucrates ductor*	Widespread	Pelagic
Perciformes	Carangidae	Focal species	*Seriola dumerili*	Widespread	Benthic/pelagic
Perciformes	Carangidae	Focal species	*Seriola hippos*	Australasian	Benthic/pelagic
Perciformes	Carangidae	Focal species	*Seriola rivoliana*	Widespread	Benthic/pelagic
Perciformes	Carangidae	NZ native	*Decapterus koheru*		
Perciformes	Carangidae	NZ native	*Pseudocaranx georgianus*		
Perciformes	Carangidae	NZ native	*Seriola lalandi*		
Perciformes	Carangidae	NZ native	*Trachurus declivis*		
Perciformes	Carangidae	NZ native	*Trachurus murphyi*		
Perciformes	Carangidae	NZ native	*Trachurus novaezelandiae*		
Perciformes	Chaetodontidae	Focal species	*Amphichaetodon howensis*	Australasian	Benthic
Perciformes	Chaetodontidae	Focal species	*Chaetodon guentheri*	Tropical SW Pacific	Benthic
Perciformes	Chaetodontidae	Focal species	*Forcipiger flavissimus*	Indo-Pacific	Benthic
Perciformes	Cheilodactylidae	Focal species	*Morwong ephippium*	Australasian	Benthic
Perciformes	Cheilodactylidae	Focal species	*Morwong fuscus*	Australasian	Benthic
Perciformes	Cheilodactylidae	Focal species	*Pseudogoniistius nigripes*	Australasian	Benthic
Perciformes	Cheilodactylidae	NZ native	*Cheilodactylus spectabilis*		
Perciformes	Cheilodactylidae	NZ native	*Nemadactylus douglasii*		
Perciformes	Cheilodactylidae	NZ native	*Nemadactylus macropterus*		
Perciformes	Cheilodactylidae	NZ native	*Nemadactylus n. sp.*		
Perciformes	Cirrhitidae	Focal species	*Cyprinocirrhites polyactis*	Indo-Pacific	Benthic

Anguilliformes	Congridae	Focal species	*Gorgasia japonica*	Tropical SW Pacific	Benthic
Anguilliformes	Congridae	NZ native	*Bassanago bulbiceps*		
Anguilliformes	Congridae	NZ native	*Bassanago hirsutus*		
Anguilliformes	Congridae	NZ native	*Conger monganius*		
Anguilliformes	Congridae	NZ native	*Conger verreauxi*		
Anguilliformes	Congridae	NZ native	*Gnathophis habenatus*		
Anguilliformes	Congridae	NZ native	*Gnathophis umbrellabius*		
Anguilliformes	Congridae	NZ native	*Scalanago lateralis*		
Perciformes	Coryphaenidae	Focal species	*Coryphaena hippurus*	Widespread	Pelagic
Scorpaeniformes	Dactylopteridae	Focal species	*Dactyloptena orientalis*	Widespread	Benthic
Tetraodontiformes	Diodontidae	Focal species	*Chilomycterus reticulatus*	Widespread	Benthic/pelagic
Tetraodontiformes	Diodontidae	Focal species	*Diodon hystrix*	Widespread	Benthic
Tetraodontiformes	Diodontidae	NZ native	*Allomycterus pilatus*		
Perciformes	Echeneidae	Focal species	*Echeneis naucrates*	Widespread	Pelagic
Perciformes	Echeneidae	Focal species	*Phtheirichthys lineatus*	Widespread	Pelagic
Perciformes	Echeneidae	Focal species	*Remora albescens*	Widespread	Pelagic
Perciformes	Echeneidae	Focal species	*Remora brachyptera*	Widespread	Pelagic
Perciformes	Echeneidae	Focal species	*Remora osteochir*	Widespread	Pelagic
Perciformes	Echeneidae	Focal species	*Remora remora*	Widespread	Pelagic
Perciformes	Ephippidae	Focal species	*Platax tiera*	Widespread	Benthic/pelagic
Sygnathiformes	Fistulariidae	Focal species	*Fistularia commersonii*	Indo-Pacific	Benthic
Sygnathiformes	Fistulariidae	Focal species	*Fistularia petimba*	Widespread	Benthic
Perciformes	Girellidae	Focal species	*Girella cyanea*	Australasian	Benthic
Perciformes	Girellidae	Focal species	*Girella fimbriata*	Endemic	Benthic
Perciformes	Girellidae	NZ native	*Girella tricuspidata*		
Perciformes	Gobiidae	Focal species	*Acentrogobius pflaumii*	Tropical SW Pacific	Benthic
Perciformes	Gobiidae	Focal species	*Arenigobius bifrenatus*	Australasian	Benthic
Perciformes	Gobiidae	Focal species	*Gobiopterus semivestitus*	Australasian	Benthic
Perciformes	Gobiidae	Focal species	*Valenciennea helsdingenii*	Indo-Pacific	Benthic

Perciformes	Gobiidae	NZ native	*Favonigobius exquisitus*		
Perciformes	Gobiidae	NZ native	*Favonigobius lentiginosus*		
Perciformes	Gobiidae	NZ native	*Gobiopsis atrata*		
Gonorynchiformes	Gonorynchidae	Focal species	*Gonorynchus greyi*	Tropical SW Pacific	Benthic
Gonorynchiformes	Gonorynchidae	NZ native	*Gonorynchus forsteri*		
Perciformes	Grammistidae	Focal species	*Aulacocephalus temmincki*	Indo-Pacific	Benthic
Perciformes	Kyphosidae	Focal species	*Kyphosus bigibbus*	Indo-Pacific	Benthic
Perciformes	Kyphosidae	Focal species	*Kyphosus sectatrix*	Widespread	Benthic
Perciformes	Kyphosidae	Focal species	*Kyphosus vaigiensis*	Widespread	Benthic
Perciformes	Kyphosidae	NZ native	*Kyphosus sydneyanus*		
Perciformes	Labridae	Focal species	*Anampses elegans*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Bodianus bimaculatus*	Indo-Pacific	Benthic
Perciformes	Labridae	Focal species	*Bodianus flavifrons*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Bodianus flavipinnus*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Bodianus izuensis*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Bodianus masudai*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Coris aygula*	Indo-Pacific	Benthic
Perciformes	Labridae	Focal species	*Coris bulbifrons*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Coris dorsomacula*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Coris picta*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Leptoscarus vaigiensis*	Unknown	Benthic
Perciformes	Labridae	Focal species	*Notolabrus inscriptus*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Odax cyanoallix*	Endemic	Benthic
Perciformes	Labridae	Focal species	*Oxycheilinus bimaculatus*	Indo-Pacific	Benthic
Perciformes	Labridae	Focal species	*Pseudojuloides elongatus*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Pseudolabrus luculentus*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Suezichthys arquatus*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Suezichthys aylingi*	Australasian	Benthic
Perciformes	Labridae	Focal species	*Thalassoma amblycephalum*	Tropical SW Pacific	Benthic

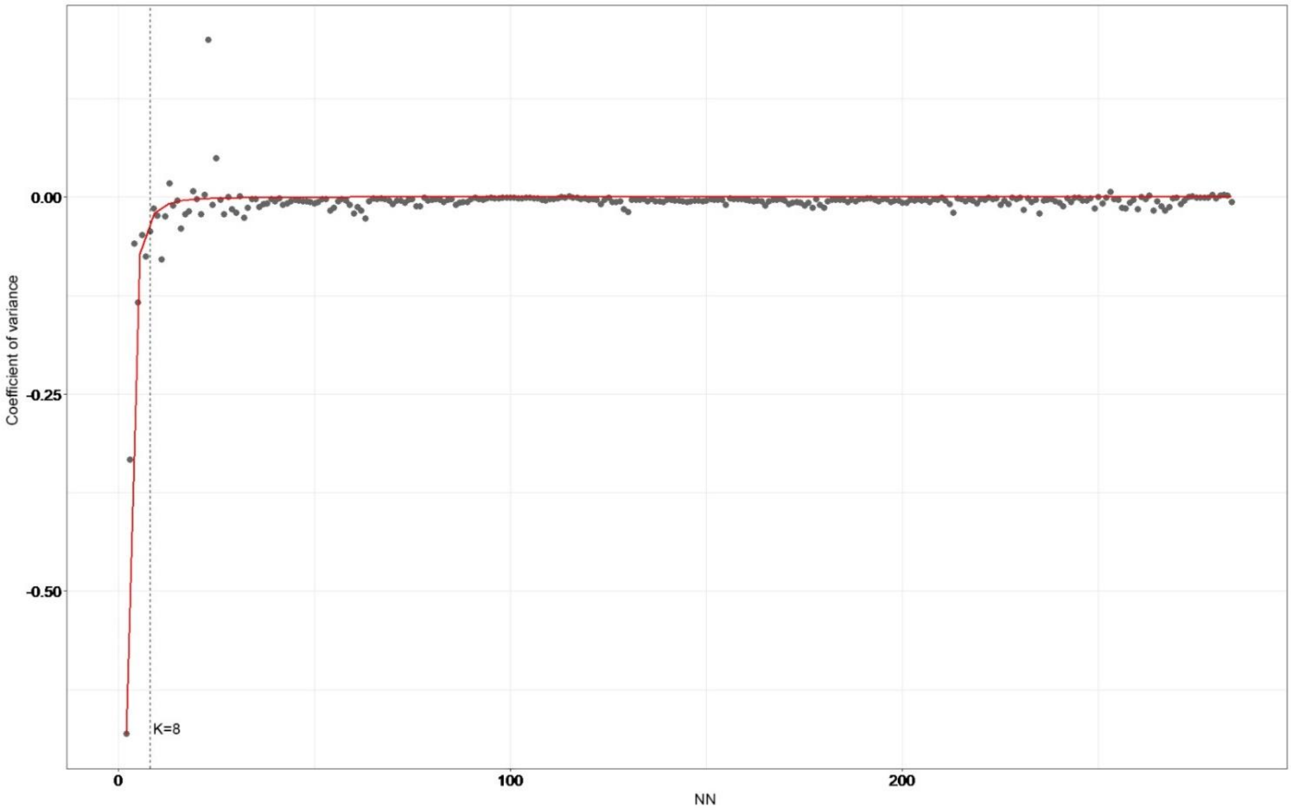
Perciformes	Labridae	Focal species	*Thalassoma lunare*	Tropical SW Pacific	Benthic
Perciformes	Labridae	Focal species	*Thalassoma lutescens*	Tropical SW Pacific	Benthic
Perciformes	Labridae	NZ native	*Bodianus unimaculatus*		
Perciformes	Labridae	NZ native	*Coris sandeyeri*		
Perciformes	Labridae	NZ native	*Notolabrus celidotus*		
Perciformes	Labridae	NZ native	*Notolabrus cinctus*		
Perciformes	Labridae	NZ native	*Notolabrus fucicola*		
Perciformes	Labridae	NZ native	*Odax pullus*		
Perciformes	Labridae	NZ native	*Pseudolabrus miles*		
Perciformes	Lutjanidae	Focal species	*Etelis carbunculus*	Unknown	Benthic
Perciformes	Malacanthidae	Focal species	*Malacanthus brevirostris*	Indo-Pacific	
Perciformes	Veliferidae	Focal species	*Metavelifer multiradiatus*	Widespread	Benthic
Perciformes	Microcanthidae	Focal species	*Atypichthys latus*	Australasian	Benthic
Perciformes	Microdesmidae	Focal species	*Parioglossus marginalis*	Tropical SW Pacific	Benthic
Tetraodontiformes	Molidae	Focal species	*Masturus lanceolatus*	Widespread	Pelagic
Tetraodontiformes	Molidae	Focal species	*Mola tecta*	Unknown	Pelagic
Tetraodontiformes	Molidae	Focal species	*Ranzania laevis*	Widespread	Pelagic
Tetraodontiformes	Molidae	NZ native	*Mola ramsayi*	Widespread	Pelagic
Tetraodontiformes	Monacanthidae	Focal species	*Aluterus monoceros*	Widespread	Benthic/pelagic
Tetraodontiformes	Monacanthidae	Focal species	*Aluterus scriptus*	Widespread	Benthic
Tetraodontiformes	Monacanthidae	Focal species	*Cantherhines dumerili*	Indo-Pacific	Benthic
Tetraodontiformes	Monacanthidae	Focal species	*Nelusetta ayraud*	Australasian	Pelagic
Tetraodontiformes	Monacanthidae	Focal species	*Thamnaconis analis*	Australasian	Benthic
Tetraodontiformes	Monacanthidae	NZ native	*Meuschenia scaber*		
Tetraodontiformes	Monacanthidae	Focal species	*Monocentris japonica*	Indo-Pacific	Benthic
Perciformes	Mullidae	Focal species	*Parapeneus spilurus*	Indo-Pacific	Benthic

Perciformes	Mullidae	Focal species	*Upeneus francisi*	Australasian	Benthic
Perciformes	Mullidae	NZ native	*Upeneichthys porosus*		
Anguilliformes	Muraenidae	Focal species	*Enchelycore ramosa*	Tropical SW Pacific	Benthic
Anguilliformes	Muraenidae	Focal species	*Gymnothorax annasona*	Australasian	Benthic
Anguilliformes	Muraenidae	Focal species	*Gymnothorax berndti*	Unknown	Benthic
Anguilliformes	Muraenidae	Focal species	*Gymnothorax obesus*	Australasian	Benthic
Anguilliformes	Muraenidae	Focal species	*Gymnothorax porphyreus*	Tropical SW Pacific	Benthic
Anguilliformes	Muraenidae	Focal species	*Gymnothorax prionodon*	Tropical SW Pacific	Benthic
Anguilliformes	Muraenidae	NZ native	*Gymnothorax nubilus*		
Anguilliformes	Muraenidae	NZ native	*Gymnothorax prasinus*		
Anguilliformes	Ophichthidae	Focal species	*Apterichthus flavicaudus*	Unknown	Benthic
Anguilliformes	Ophichthidae	Focal species	*Quassiremus polyclitellum*		
Anguilliformes	Ophichthidae	NZ native	*Ophisurus serpens*		
Anguilliformes	Ophichthidae	NZ native	*Scolecenchelys australis*		
Anguilliformes	Ophichthidae	NZ native	*Scolecenchelys breviceps*		
Anguilliformes	Ophichthidae	NZ native	*Scolecenchelys castlei*		
Perciformes	Oplegnathidae	Focal species	*Oplegnathus fasciatus*	Tropical SW Pacific	Benthic
Tetraodontiformes	Ostraciidae	Focal species	*Kentrocapros eco*	Endemic	Pelagic
Tetraodontiformes	Ostraciidae	Focal species	*Lactoria diaphana*	Indo-Pacific	Benthic
Tetraodontiformes	Ostraciidae	Focal species	*Ostracion cubicus*	Indo-Pacific	Benthic
Tetraodontiformes	Ostraciidae	Focal species	*Polyplacapros tyleri*	Australasian	Benthic
Perciformes	Pentacerotidae	Focal species	*Evistias acutirostris*	Indo-Pacific	Benthic
Perciformes	Pentacerotidae	NZ native	*Paristiopterus labiosus*		
Perciformes	Pentacerotidae	NZ native	*Pentaceros decacanthus*		
Perciformes	Pentacerotidae	NZ native	*Pentaceros richardsoni*		
Perciformes	Pentacerotidae	NZ native	*Zanclistius elevatus*		

Perciformes	Pinguipedidae	Focal species	*Parapercis binivirgata*	Unknown	Benthic
Perciformes	Pinguipedidae	NZ native	*Parapercis colias*		
Perciformes	Pinguipedidae	NZ native	*Parapercis gilliesii*		
Perciformes	Pomacentridae	Focal species	*Abudefduf vaigiensis*	Tropical SW Pacific	Benthic
Perciformes	Pomacentridae	Focal species	*Chromis abyssicola*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Chromis flavomaculata*	Tropical SW Pacific	Benthic
Perciformes	Pomacentridae	Focal species	*Chromis fumea*	Tropical SW Pacific	Benthic
Perciformes	Pomacentridae	Focal species	*Chromis hypsilepis*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Chromis vanderbilti*	Tropical SW Pacific	Benthic
Perciformes	Pomacentridae	Focal species	*Chrysiptera notialis*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Chrysiptera rapanui*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Parma kermadecensis*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Parma polylepis*	Australasian	Benthic
Perciformes	Pomacentridae	Focal species	*Stegastes gascoynei*	Australasian	Benthic
Perciformes	Pomacentridae	NZ native	*Chromis dispilus*		
Perciformes	Pomacentridae	NZ native	*Parma alboscapularis*		
Perciformes	Pomatomidae	Focal species	*Pomatomus saltatrix*	Widespread	Pelagic
Perciformes	Scombridae	Focal species	*Acanthocybium solandri*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Allothunnus fallai*	Indo-Pacific	Pelagic
Perciformes	Scombridae	Focal species	*Auxis rochei*	Widespread	Pelagic
Perciformes	Scombridae	Focal species	*Auxis thazard*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Gasterochisma melampus*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Katsuwonus pelamis*	Widespread	Pelagic
Perciformes	Scombridae	Focal species	*Sarda australis*	Australasian	Pelagic
Perciformes	Scombridae	NZ native	*Scomber australasicus*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Thunnus alalunga*	Widespread	Pelagic

Perciformes	Scombridae	NZ native	*Thunnus albacares*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Thunnus maccoyii*	Indo-Pacific	Pelagic
Perciformes	Scombridae	NZ native	*Thunnus obesus*	Widespread	Pelagic
Perciformes	Scombridae	NZ native	*Thunnus orientalis*	Widespread	Pelagic
Scorpaeniformes	Scorpaenidae	Focal species	*Maxillicosta raoulensis*	Australasian	Benthic
Scorpaeniformes	Scorpaenidae	Focal species	*Pterois volitans*	Indo-Pacific	Benthic
Scorpaeniformes	Scorpaenidae	Focal species	*Scorpaenodes evides*	Tropical SW Pacific	Benthic
Scorpaeniformes	Scorpaenidae	NZ native	*Scorpaena cardinalis*		
Scorpaeniformes	Scorpaenidae	NZ native	*Scorpaena onaria*		
Scorpaeniformes	Scorpaenidae	NZ native	*Scorpaena papillosa*		
Perciformes	Scorpididae	Focal species	*Bathystethus cultratus*	Australasian	Benthic/pelagic
Perciformes	Scorpididae	Focal species	*Labracoglossa nitida*	Australasian	Benthic/pelagic
Perciformes	Scorpididae	NZ native	*Scorpis lineolata*		
Perciformes	Scorpididae	NZ native	*Scorpis violacea*		
Perciformes	Serranidae	Focal species	*Acanthistius cinctus*	Australasian	Benthic
Perciformes	Serranidae	Focal species	*Epinephelus daemelii*	Australasian	Benthic
Perciformes	Serranidae	Focal species	*Epinephelus lanceolatus*	Indo-Pacific	Benthic
Perciformes	Serranidae	Focal species	*Epinephelus rivulatus*	Tropical SW Pacific	Benthic
Perciformes	Serranidae	Focal species	*Hypoplectrodes* sp. A	Unknown	Benthic
Perciformes	Serranidae	Focal species	*Hyporthodus octofasciatus*	Widespread	Benthic
Perciformes	Serranidae	Focal species	*Trachypoma macracanthus*	Tropical SW Pacific	Benthic
Perciformes	Serranidae	NZ native	*Caesioperca lepidoptera*		
Perciformes	Serranidae	NZ native	*Caprodon longimanus*		
Perciformes	Serranidae	NZ native	*Hypoplectrodes huntii*		
Perciformes	Serranidae	NZ native	*Hypoplectrodes* sp. B		
Perciformes	Serranidae	NZ native	*Lepidoperca aurantia*		
Perciformes	Serranidae	NZ native	*Lepidoperca tasmanica*		
Pleuronectiformes	Soleidae	Focal species	*Aseraggodes bahamondei*	Widespread	Benthic

Perciformes	Sparidae	Focal species	*Acanthopagrus australis*	Tropical SW Pacific	Benthic
Perciformes	Sparidae	Focal species	*Acanthopagrus pacificus*	Tropical SW Pacific	Benthic
Perciformes	Sparidae	NZ native	*Chrysophrys auratus*		
Perciformes	Sphyraenidae	Focal species	*Sphyraena acutipinnis*	Indo-Pacific	Pelagic
Perciformes	Sphyraenidae	Focal species	*Sphyraena qenie*	Widespread	Benthic
Perciformes	Sphyraenidae	Focal species	*Sphyraena waitii*	Australasian	Pelagic
Perciformes	Synodontidae	Focal species	*Synodus doaki*	Tropical SW Pacific	Benthic
Perciformes	Synodontidae	Focal species	*Synodus similus*	Tropical SW Pacific	Benthic
Perciformes	Synodontidae	Focal species	*Synodus variegatus*	Indo-Pacific	Benthic
Perciformes	Synodontidae	Focal species	*Trachinocephalus trachinus*	Australasian	Benthic
Tetraodontiformes	Tetraodontidae	Focal species	*Arothron firmamentum*	Widespread	Pelagic
Tetraodontiformes	Tetraodontidae	Focal species	*Arothron stellatus*	Widespread	Benthic
Tetraodontiformes	Tetraodontidae	Focal species	*Canthigaster callisterna*	Australasian	Benthic
Tetraodontiformes	Tetraodontidae	Focal species	*Lagocephalus cheesmanii*	Australasian	Pelagic
Tetraodontiformes	Tetraodontidae	Focal species	*Lagocephalus lagocephalus*	Widespread	Pelagic
Tetraodontiformes	Tetraodontidae	Focal species	*Lagocephalus sceleratus*	Indo-Pacific	Benthic
Tetraodontiformes	Tetraodontidae	Focal species	*Pelagocephalus marki*	Widespread	Pelagic
Tetraodontiformes	Tetraodontidae	NZ native	*Contusus richei*		
Tetraodontiformes	Tetraodontidae	NZ native	*Sphoeroides pachygaster*		
Perciformes	Zanclidae	Focal species	*Zanclus cornutus*	Tropical SW Pacific	Benthic



FigureS1.1: The coefficient of variation for the mean for Euclidean straight-line distance between spatial locations of focal species occurrences for different values of K. By K=8 there is a marked flattening of the variance curve. However, due to the distribution and clustering of our data and the geography of New Zealand's coastline, the mean spatial distance at K=8 (22km) was too high for us to make inferences of spatial patterns at appropriate scales. Hence, K=4 was the chosen value for K.

Table S1.2: Total number of occurrences for each biogeographic affinity. We used the categories defined by Roberts et al., 2015 (Roberts, C., Stewart, A. L., & Struthers, C. D. (2015). The Fishes of New Zealand (C. D. Roberts, A. L. Stewart, & C. D. Struthers Eds.): Te Papa Press).

Affinity	Number of occurrences
Endemic	16
Unknown	31
Indo-Pacific	173
Widespread	195
Tropical and Southwestern Pacific	360
Australasian	882

2.7.2 Supplement 2: New-to-New Zealand fish occurrences metadata and global distributions

Table S2.1: New records of marine fishes in the New Zealand, and/or New Zealand mainland contributed by this study. [1]Denotes species that had previously been recorded only from Rangitāhua/Kermadec Islands. Locations are abbreviated as follows: FN, Far North; HCI, Hen and Chickens Islands, HG, Hauraki Gulf; PKI, Poor Knights Islands. Estimated size of the fish is total length, TL. [2]Pomatomidae are a new-to-New Zealand family. [3]Provisional identification by global experts, the taxonomy of the Synodontidae of the South Pacific are currently being reviewed and the cryptic nature of the Antennaridae means that a tissue sample or specimen is needed to confirm identification. The images provided by citizen scientists that were used for new-to - New Zealand species accounts are included below the table.

Family	**Species**	**Who provided the ID**	**Rationale for ID**	**Date**	**Affinity**	**Location and coordinates**	**Depth (m)**	**Size (TL, mm)**
Labridae	*Bodianus bimaculatus* Allen 1973	Malcolm Francis	Body depth and blunt snout, orange colouration with 6 narrow red/pink stripes, yellow-edged black spot on operculum and a black spot on caudal peduncle.	30-Sep-01	Indo-Pacific	Taravana cave, PKI (-35.4653, 174.7361)		90
Labridae	*Oxycheilinus bimaculatus* Valenciennes 1840	Malcom Francis	Dorsal profile of head slightly convex, dorsal rays IX 10, Caudal fin rounded, dark blotch about pectoral fin followed by three less distinct blotches posteriorly, green spot on the first dorsal fin membrane.	18-Apr-12	Indo-Pacific	Jan's Tunnel, PKI (-35.4883, 174.7361)		150
Pomatomidae[2]	*Pomatomus saltatrix* Linnaeus 1766	Primary author	Separate dorsal fins, dorsal fin VII-VIII + I, 23-28, forked tail, jaws extend beyond eye, single row of teeth, protruding lower jaw.	8-Dec-12	Widespread	Cavalli Islands, FN (-34.9979, 173.9647)		450
Oplegnathidae	*Oplegnathus fasciatus* Temminck & Schlegel, 1844	Clinton Duffy	Dorsal rays XI-XII, 17-18, 6-7, grey colouration with broad vertical black bands, beak-like jaws.	17-Nov-13	Tropical southwestern Pacific	Leigh, HG (-36.2779, 174.8238)	17	270
Muranidae	*Gymnothorax annasona* Whitley 1937	Andrew Stewart	Mottled yellowish moray with pale grey head and numerous larger dark blotches, yellow eye and white margin on dorsal fins	14-Apr-14	Australasian	Barren Arch, PKI (-35.4812, 174.7368)	10	1000+
Tetraodontidae	*Lagocephalus sceleratus* Gmelin 1789	Andrew Stewart	Distinctive silver stripe along the sides of the body and greenish colour dorsally with dark spots. Two lateral lines, front of head bluntly rounded.	4-Jul-15	Indo-Pacific	Omaha Bay, HG (-36.3299, 174.8363)		220
Monocanthidae	*Cantherhines dumerilii*[1] Hollard 1854	Tom Trnski	Prominent first dorsal fin, dorsal rays II, 34-39. Juvenile colouration: light grey with small white spots. Yellow iris and yellow hue to fins. Two very small spines on the side of the tail base	Jan-16	Indo-Pacific	Three Kings Islands, FN (-34.1729, 172.0973	2	220

Labridae	*Bodianus izuensis* Araga & Yoshino 1975	Primary author	Consistent with colouration in Gomon (2006)[1]. Two or three moderately broad black stripes on body; black spot on scaly caudal-fin base, if present, indistinguishable from second black stripe	27-Apr-17	Tropical southwestern Pacific	Serpent Rock, PKI (-35.4618, 174.7416)	18	75
Labridae	*Bodianus masudai* Araga & Yoshino 1975	Tom Trnski	Consistent with colouration in Gomon (2006) [1]. Specifically: Dorsal fin margin with white tips; dorsolateral stripe yellow; lateral and ventrolateral stripe pale yellow to white; dark spot on operculum margin; caudal fin blackish separated medially with red stripe, and caudal tip white	3-Feb-19	Tropical southwestern Pacific	Northern Arch, PKI (-35.4480, 174.7315)	54	75
Antennaridae	*Antennarius commerson* Lacépède 1798	Ted Pietsch	Illicium about twice as long as second dorsal spine; esca with fleshy tapering tip with basal cluster of short filaments; second and base of third dorsal spines joined by thick membrane[3]	1-Mar-19	Indo-Pacific	Maroro Bay, PKI (-35.4618, 174.7416)	11	120
Synodontidae	*Synodus variegatus*[1] Lacépède 1803	Barry Russell	Flap on anterior nostril short and triangular; irregular bars dorsally on body; whitish stripe ventrolaterally on body interrupted by reddish bars; lacks prominent black spot on base of caudal fin[3]	18-Mar-19	Widespread	Coppermine island, HCI (-35.9232, 175.1246)	10	170
Zanclidae	*Zanclus cornutus*[1] Linnaeus 1758	Primary author	Compressed discoid body, tubular snout, elongated sickle shaped dorsal spines, two broad black bars on body and yellow saddle across snout	14-Mar-19	Tropical southwestern Pacific	Ngaio Rock, PKI (-35.4896, 174.7381)	25	250
Blenniidae	*Parablennius intermedius* Ogilby 1915	Tom Trnski	Dorsal fin XII,17; long tentacles above eye with multiple lobes; small spots on head, larger dark spots along body	27-Mar-19	Tropical southwestern Pacific	Parengarenga, FN (-34.5346, 172.9440)	7	90
Blenniidae	*Plagiotremus rhinorhynchos* Bleeker 1852	Primary author	Dorsal rays XI, 32-36, small pelvic fins, Blue colouration with two narrow bright blue lines running the length of body from the tip of the snout to caudal fin base, lacking the black spots or bars of congener *P.tapeinosoma*	27-Apr-19	Indo-Pacific	Ngaio rock, PKI (-35.4890, 174.7387)	8	120
Synodontidae	*Trachinocephalus trachinus* Temminck & Schlegel 1846	Tom Trnski	Very short snout and an oblique mouth with numerous fine, sharp teeth. Narrow pale blue and yellow wavy stripes and several indistinct darker bars along the sides, and a black spot at the upper corner of the gill opening. *T.trachinus* is the only Trachinocephalus species distributed in the Indo-West Pacific Ocean	1-May-20	Indo-Pacific	Houhora Harbour, FN (-34.7965, 173.1200)	12	230
Chaetodontidae	*Chaetodon guentheri*[1]Ahl 1923	Tom Trnski	White to pale yellow colouration with small dark spots that form oblique or horizontal lines. The rear part of the body, soft dorsal fin and anal fin are yellow. Dorsal and anal fins	20-May-20	Tropical southwestern Pacific	Maroro Bay, PKI (-35.4884, 174.7395)	15	70

			with a narrow white margin and a black submarginal band. A black bar passes through the eye.					
Gobiidae	*Valenciennea helsdingenii* Bleeker 1858	Primary author	White colouration with two brown to orange stripes running the length of the body extending onto the head with lower line extending onto upper jaw, prominent black dot with while margin on first dorsal fin.	22-April-21	Indo-Pacific	Maunganui Bay / Deep Water Cove, BOI (-35.1936, 174.2986)	14	70

Antennarius commerson, Maroro Bay

Bodianus bimaculatus, Jan's Tunnel

Bodianus izuensis, Serpent rock

Bodianus masudai, Northern Arch

Cantherhines dumerilii, Three Kings Islands

Chaetodon guentheri, Maroro Bay

Gymnothorax annasona, Maroro Bay

Lagocephalus sceleratus, Omaha Bay

Oplegnathus fasciatus, Leigh Coast

Oxycheilinus bimaculatus, Jan's Tunnel

Parablennius intermedius, Parengarenga Hbr

Plagiotremus rhinorhynchos, South Harbour

Synodus variegatus, Hen & Chickens Islands

Trachinocephalus trachinus, Houhora Harbour

Pomatomus saltatrix, Cavalli Islands

Valenciennea helsdingenii, Deepwater Cove

Zanclus cornutus, South Harbour

Figure S2.1 Images submitted by the citizen scientists as an account of the new species observations. The image of *Bodianus bimaculatus* that was used as evidence of the occurrence was held in Malcolm Francis collections and a stock image was substituted from Izuzuki / http://www.izuzuki.com/, License: CC by Attribution-NonCommercial-ShareAlike

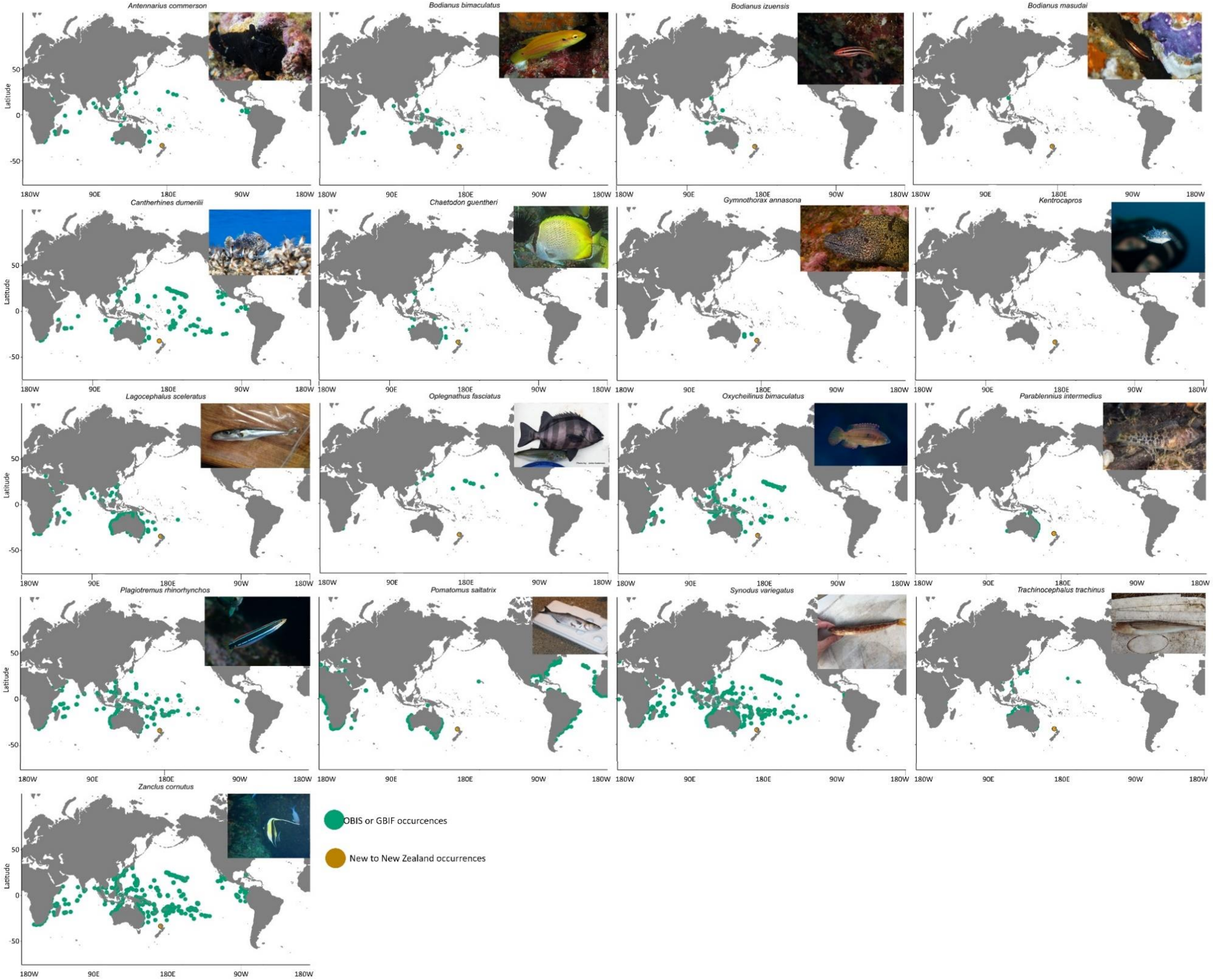


Figure S2.2: Global occurrence data for all new-to-New Zealand and new-to-New Zealand mainland fishes (green points) and the New Zealand record (yellow points). Occurrence data for *Trachinocephalus trachinus* was sourced from GBIF (2020). Data for all other species sourced from OBIS (2020).

Images are those submitted by the citizen scientists as an account of the new species observations. The image of *Bodianus bimaculatus* that was used as evidence of the occurrence was held in Malcolm Francis collections and a stock image was substituted from Izuzuki / http://www.izuzuki.com/, License: CC by Attribution-NonCommercial-ShareAlike

References:

GBIF. (2020). GBIF.org (22 June 2020) GBIF Occurrence Download https://doi.org/10.15468/dl.234wva

OBIS. (2020). Ocean Biodiversity Information System. Intergovernmental Oceanographic Commission of UNESCO. www.obis.org.

2.7.3 Supplement 3: Spatial hotspots of species richness

Species richness represents the number of different focal species present clustered by 5km for each variable. Species richness is depicted by century, season, life-stage and habitat.

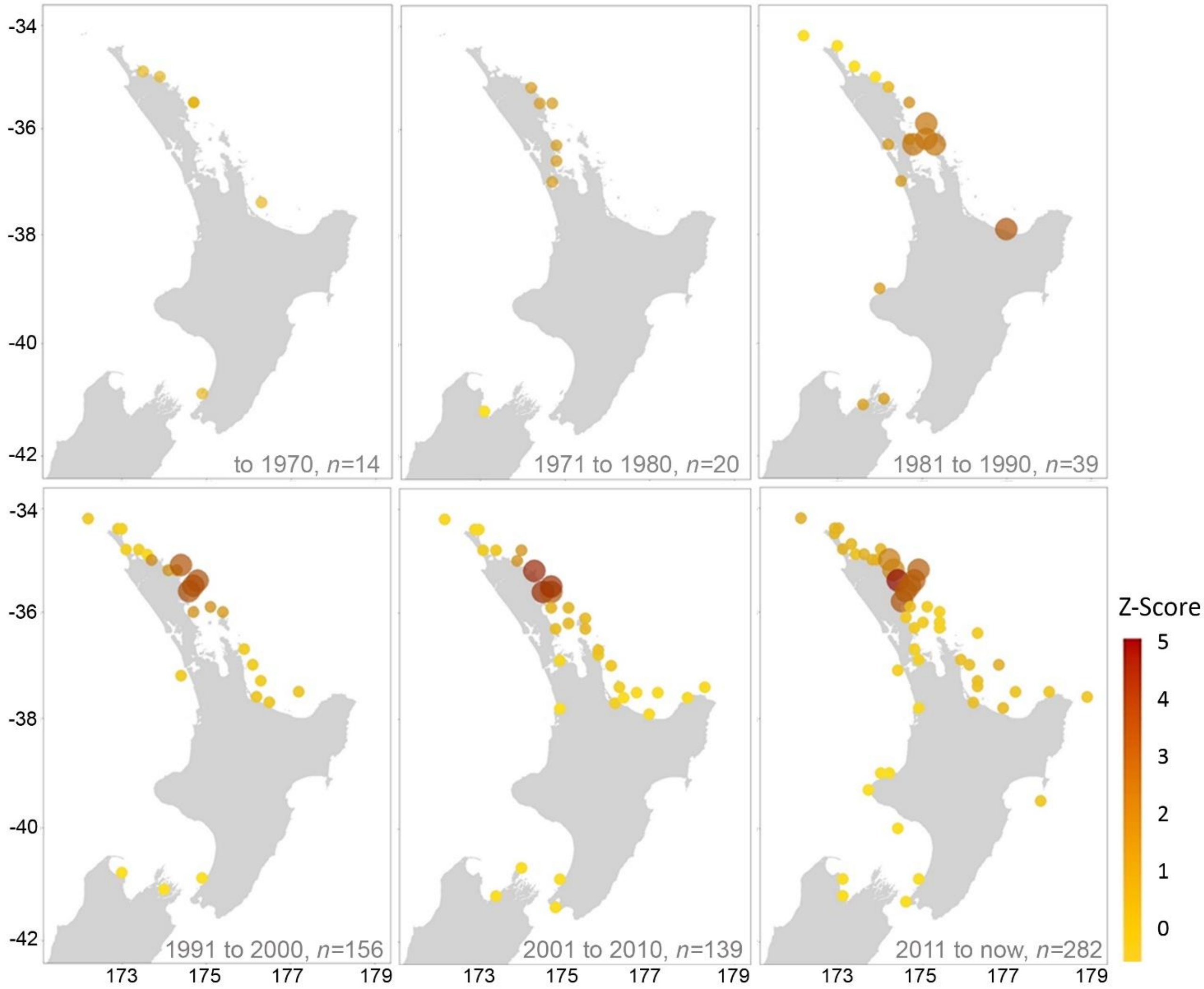


Figure S3.1: Species richness hotspots by decade. Species richness represents the number of different focal species present clustered by 5km. The colour gradient indicates the z-score, where darker colours denote locations with a high diversity of focal species compared to all other locations. Larger points are statistically significant ($p \leq 0.10$) clusters of high species richness determined using are the Getis-Ord Gi*statistic. n is the total number of distinct focal species for all locations per decade.

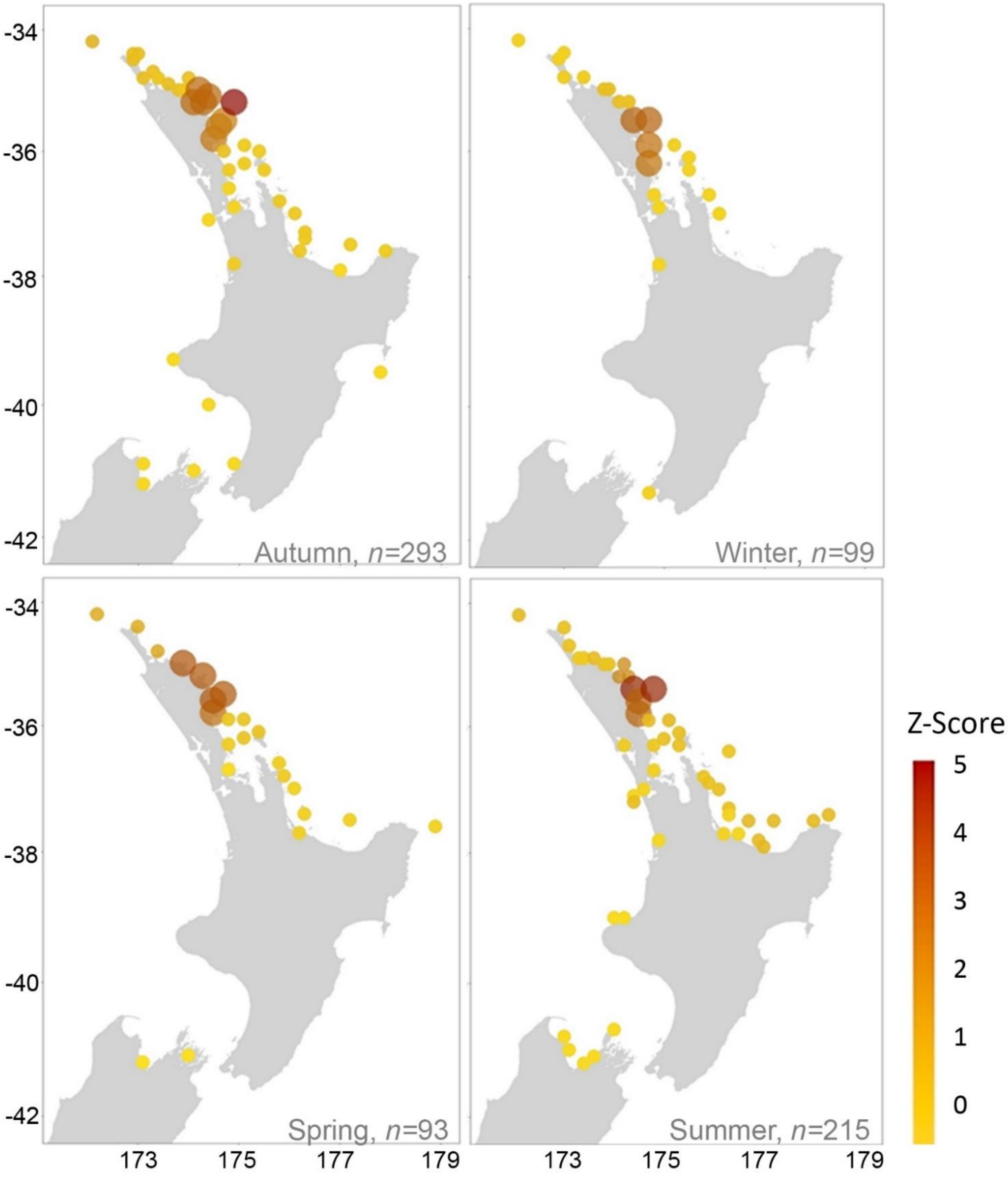


Figure S3.2: Species richness hotspots by season (Austal summer: December to February, Autumn: March to May, Winter: June to August, Spring: September to November). Species richness represents the number of different focal species present clustered by 5km. The colour gradient indicates the z-score, where darker colours denote locations with a high diversity of focal species compared to all other locations. Larger points are statistically significant ($p \leq 0.10$) clusters of high diversity determined using are the Getis-Ord Gi*statistic. n is the total number of distinct focal species for all locations per location per season.

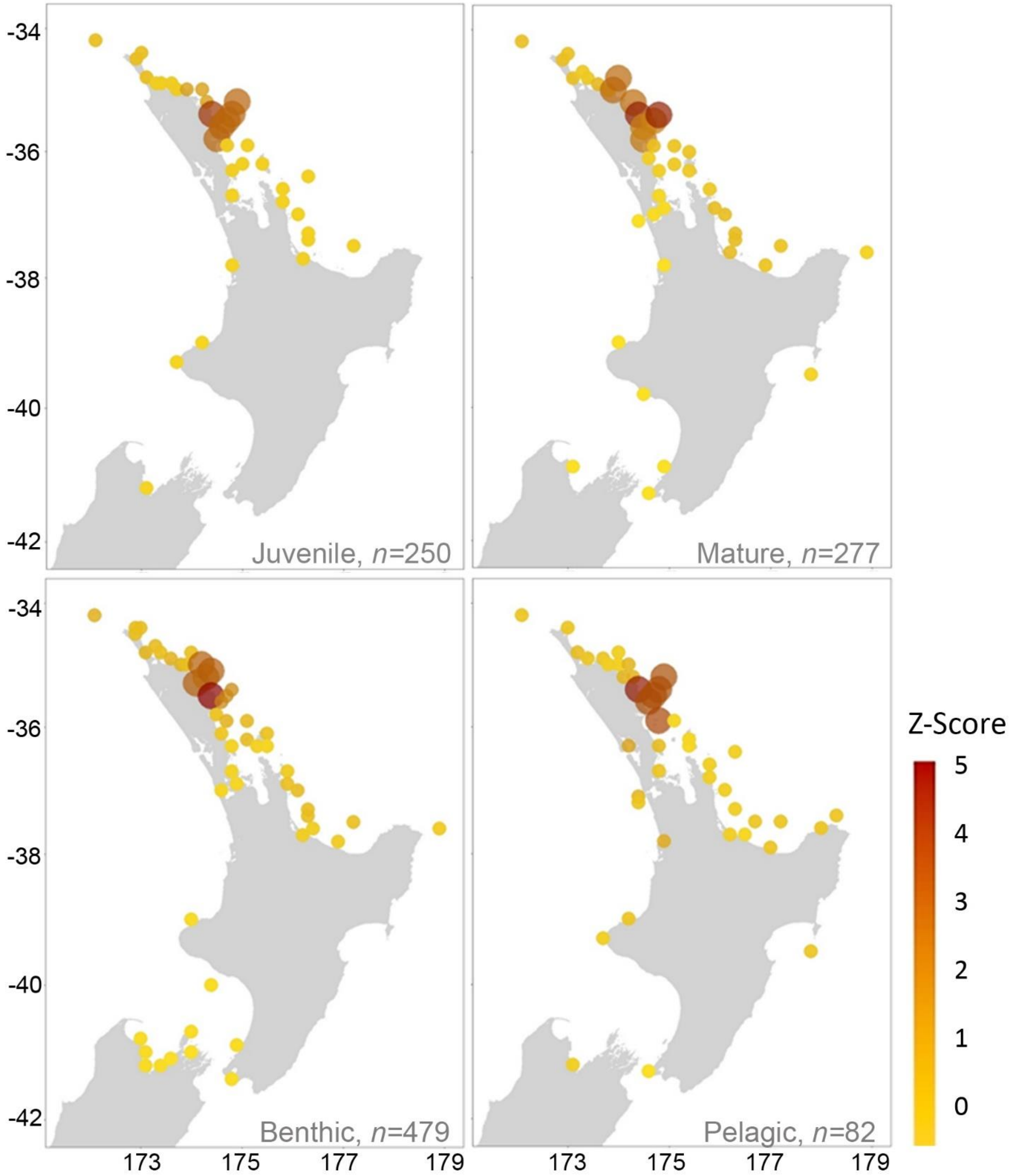


Figure S3.3: Species richness hotspots by life-stage (Juvenile or Mature) and Habitat (Pelagic and Benthic). Species richness represents the number of different focal species present clustered by 5km. The colour gradient indicates the z-score, where darker colours denote locations with a high diversity of focal species compared to all other locations. Larger points are statistically significant ($p \leq 0.10$) clusters of high diversity determined using are the Getis-Ord Gi*statistic. n is the total number of distinct focal species for all locations per location for each life-stage or habitat.

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STATEMENT OF CONTRIBUTION
DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

We, the candidate and the candidate's Primary Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of candidate:	Irene Middleton
Name/title of Primary Supervisor:	Dr Libby Liggins
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Middleton, I., Aguirre, J. D., Trnski, T., Francis, M., Duffy, C., & Liggins, L. (2021). Introduced alien, range extension or just visiting? Combining citizen science observations and expert knowledge to classify range dynamics of marine fishes. Diversity and Distributions, n/a(n/a) https://doi.org/10.1111/ddi.13273

○ The manuscript is currently under review for publication – please indicate:

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- Describe the contribution that the candidate has made to the manuscript/published work:

Conceptual design, Facilitated workshops, collected the data, developed decision frameworks and analysed the data with the help of supervisors, drafted the manuscript, made all graphs and figures.

○ It is intended that the manuscript will be published, but it has not yet been submitted to a journal

Candidate's Signature:	Irene Middleton Digitally signed by Irene Middleton Date: 2021.11.12 14:38:01 +13'00'
Date:	12-Nov-2021
Primary Supervisor's Signature:	Libby Liggins Digitally signed by Libby Liggins DN: cn=Libby Liggins, c=NZ, o=Massey University, ou=School of Natural and Computational Sciences, email=l.liggins@massey.ac.nz Date: 2021.11.14 [illegible] +13'00'
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DRC 19/09/10

3 Introduced alien, range extension or just visiting? Combining citizen science observations and expert knowledge to classify range dynamics of marine fishes

3.1 Abstract

Aim: Despite the unprecedented rate of species redistribution during the Anthropocene, there are few monitoring programs at the appropriate spatial and temporal scale to detect distributional change of marine species, and to infer climate- versus human-mediated drivers of change. Here, we present an approach that combines citizen science with expert knowledge to classify out-of-range occurrences for marine fishes as potential range extensions or human-mediated dispersal events.

Innovation: Our stepwise approach includes decision trees, scoring, and matrices to classify citizen science observations of species occurrences and to provide a measure of confidence and validation using expert knowledge. Our method draws on peer-reviewed literature, knowledge of the species (e.g., contributing to its detectability, and potential to raft with, or foul, man-made structures or debris), and information obtained from citizen science observations (e.g., life-stage, number of individuals). Using a case-study of suspected out-of-range marine fishes in Aotearoa New Zealand, we demonstrate our approach to defining species' ranges, assigning confidence to these definitions, and considering the species detectability to overcome the data deficiencies that currently hinder monitoring the range dynamics of these species. Our classification of citizen science observations revealed that six-of-ten species had out-of-range occurrences; one of these was classified as an extralimital vagrant, four species had potentially extended their ranges, and one species occurrence was likely due to human-mediated dispersal.

Conclusion: The case-study of marine fishes in New Zealand validates our approach combining citizen science observations with expert knowledge to infer species range dynamics in real-time. Our stepwise approach helps to identify data deficiencies important in informing

scientific inferences and management actions and can be refined to suit other data sources, taxonomic groups, geographic settings, or extended with new steps and existing tools.

3.2 Introduction

The rate of species redistribution during the Anthropocene is unprecedented (Chen, Hill, Ohlemüller, Roy, & Thomas, 2011) particularly in the marine environment (Poloczanska et al., 2013; Sorte, Williams, & Carlton, 2010). The primary drivers of species movement around the globe are human-mediated introductions (Walther et al., 2009; Sorte et al., 2010) and climate change (Bellard, Bertelsmeier, Leadley, Thuiller, & Courchamp, 2012; Pecl et al., 2017). In the ocean, the vectors of human-mediated movement of 'alien' species (i.e., novel or non-native species introduced outside their past or present distribution and natural dispersal potential) include biofouling and ballast water associated with shipping, and intentional or unintentional releases through the aquarium trade (e.g., Currie & Parry, 1999; Hewitt et al., 2004). Climate-mediated movement refers to 'range extending' species that are able to disperse and survive in new locations owing to changes in ocean circulation and temperatures (e.g., Sagarin, Barry, Gilman, & Baxter, 1999; Stuart-Smith et al., 2018; Vergés et al., 2014). Human-mediated introductions of 'invasive' or 'pest' species (i.e., alien species that become problematic in their new location) have had devastating impacts on recipient environments and economies (Sala et al., 2000), and although climate-mediated range extensions do not yet have the same notoriety, they pose similar risks (Johnson et al., 2011; Sorte et al., 2010). For instance, several species undergoing range extensions have already had negative impacts on recipient ecosystems – competing with native species (Arrontes, 2002) and modifying community structure (Holbrook, Schmitt, & Stephens, 1997; Ling, 2008; Madin et al., 2012) – presenting environmental, economic and social challenges (Johnson, Ling, Ross, Shepherd, & Miller, 2005; Pecl et al., 2017).

Given the potential for both human-mediated and climate-mediated species introductions to severely impact recipient ecosystems and local economies, considering both within one observation and risk evaluation framework would have several benefits. In particular, as the rate of climate-mediated species introductions increases, the resourcing of mitigation versus incursion responses and impact management may need to be revised. For instance, legislation regarding ballast discharge, hull fouling, and the aquarium trade (e.g., MPI Import Health Standard. Ornamental Fish and Marine invertebrates 2017) would not have any bearing on climate-mediated range-extending species (Pyke et al., 2008). Rather, a focus on the management of species impacts on the recipient ecosystem may need to be prioritised (e.g., culling of the range-extending urchin *Centrostephanus rodgersii* in Tasmania, Sanderson, Ling, Dominguez, & Johnson, 2015). Furthermore, responsible climate change adaptation of local economies and cultural practices requires an understanding of the potential risks posed by climate change (Adger, Barnett, Brown, Marshall, & O'Brien, 2013; Pecl et al., 2017) necessitating the identification and disentanglement of climate-mediated species introductions. Integral to such a unified approach is managers' ability to detect new species occurrences outside of their known range (i.e., out-of-range occurrences) and to infer the likely mode of introduction.

Confidently detecting out-of-range occurrences of species requires a thorough understanding of the geographic range that species occupy (Gledhill et al., 2015). Given that large areas of the marine realm remain under-surveyed (Hortal et al., 2015) and that marine species are often cryptic (Hubert et al., 2012), wide-ranging, and highly mobile (e.g., Block et al., 2005; Bonfil et al., 2005), our knowledge of geographic distributions is far from complete (Appeltans et al., 2012; Hortal et al., 2015). Furthermore, delineating the boundary of a species' distribution where individuals are naturally more diffusely or patchily distributed (McCarthy et al., 2013) and are therefore difficult to systematically survey (Bates et al., 2014; Feary et al., 2013) is a renowned challenge. Nonetheless, defining range boundaries is fundamental in detecting out-of-range occurrences for species undergoing human-mediated and climate-mediated dispersal and inferring subsequent range dynamics (Urban et al., 2016).

Systematic surveys of species range extents and biosecurity monitoring are costly, time-consuming, and often spatially restricted; hence, marine scientists and managers must often rely on diverse sources of knowledge (Delaney, Sperling, Adams, & Leung, 2008). For instance, mining existing datasets collected for other purposes such as fisheries data (e.g., Robinson et al., 2015) and museum collections (e.g.,Cheung, Watson, & Pauly, 2013; Perry, Low, Ellis, & Reynolds, 2005), and the use of opportunistic species occurrence records provided by the general public (e.g., Lenanton, Dowling, Smith, Fairclough, & Jackson, 2017; Liggins et al., 2020). In particular, citizen science – the participation of citizens in the collection of data or the provisioning of local knowledge for the planning, development and execution of scientific research (Eitzel et al., 2017) – has been useful in increasing temporal and spatial data resolution, overcoming a lack of baseline data and detecting out-of-range occurrences in both terrestrial (Theobald et al., 2015; e.g., The Great British Birdcount, MonarchWatch.org) and marine settings (Pecl et al., 2019; e.g., Range Extension Database and Mapping Project [Redmap] Reef Environmental Education Foundation [REEF]; European Alien Species Information Network [EASIN]). However, even though citizen science data has proven utility in contributing toward scientific (e.g., Lenanton et al., 2017; Soroye, Ahmed, & Kerr, 2018) and policy outcomes (e.g., Delaney et al., 2008; Madin et al., 2012), it often varies in completeness and has inherent biases that require careful quality control and validation (Bird et al., 2014; Isaac, Strien, August, Zeeuw, & Roy, 2014).

Qualitative assessment tools can help harmonize unstructured species occurrence data and incomplete distribution data, with the input of expert knowledge (Bates et al., 2014; Robinson et al., 2015). In particular, decision (or classification) trees have been used by regulatory authorities, scientists and resource managers across the fields of biosecurity (Drolet et al., 2016; Wotton & Hewitt, 2004), applied ecology and climate change science (Robinson et al., 2015). The intuitive, systematic routines of decision trees facilitate the interrogation of varied data, allowing researchers to apply knowledge or select the appropriate information at each step, and to implement expert verification for quality control. For example, drawing on knowledge of species’ biology and an expert ‘Technical Advisory Group’, Wotton & Hewitt (2004) used decision trees to inform post-border responses to invasive species incursions in New Zealand waters, and researchers in ocean warming hotspots in the United Kingdom (Hiscock, Southward, Tittley, & Hawkins, 2004) and south-eastern Australia (Robinson et al., 2015) have used decision trees to validate citizen science observations with the input of specific taxonomic and regional expertise to identify range extensions. There is great flexibility in how these qualitative assessments can be applied, so each decision framework can be tailored to the specific research or management questions, as well as the data, knowledge, and expertise available.

Here, we present a series of qualitative assessment tools that combine citizen science observations with expert knowledge to classify marine species occurrences as within- or out-of-range, and as potential climate-mediated range extensions or human-mediated introductions. Our approach builds on methods developed for similar purposes (Robinson et al., 2015, Pecl et al., 2019), but bridges climate change biology and invasion biology, to provide a framework suited to range-extending species as well as introduced alien species. Through the inclusion of life-stage information and the number of observed individuals, our method additionally provides a means to detect and track the arrival, as well as the establishment and population growth of range-extending species or introduced alien species. Our method is unique in considering seasonal variability in defining species’ ranges and therefore the detection of out-of-range occurrences for migratory species as a result of phenological shifts. Furthermore, rather than uniformly classifying the range status of a species, we classify individual occurrences to infer the local range dynamics of a species, increasing the spatial and temporal resolution of inferences, and providing baselines for future monitoring.

We use a case-study focussed on potentially out-of-range marine fish species in Aotearoa New Zealand to demonstrate the utility of our method. Oceanic island nations, like New Zealand, are isolated from source populations of both range extending and introduced alien species, yet they are reliant on vessel movements, the predominant vector of human-mediated dispersal. For these reasons, identifying the mode of species introduction, assessing the risk posed by climate change, and determining appropriate management actions, is particularly challenging. We demonstrate that with the input of citizen science observations and an expert panel our method can identify out-of-range occurrences of marine fishes in New Zealand, use individual occurrences to classify the local range dynamics of species and provide an indication as to whether the out-of-range occurrence has been climate- or human-mediated.

3.3 Methods

3.3.1 Overview of method

Our approach (Figure 1) is designed to classify occurrences of any target species as: within known range (WR), extralimital vagrancy (EV: juvenile individual observed outside of its known range), potential range extension (PR: mature individual observed outside of its known range; or several juvenile individuals observed outside their known range), or a range extension (RE: several mature individuals observed outside of their known range). Specifically, our approach first requires characterising the species known range (i.e., extent of the species' typical geographic distribution) including seasonal variation, if applicable (i.e., extent of the species typical geographic distribution for a given season/s) and assigning a measure of confidence for these defined spatio-temporal range boundaries (Step 1). In Step 2, a species detectability score is assigned using a method similar to that presented in Robinson et al., (2015), based on likely modes of encounter, the species abundance, and conspicuousness. Species observations obtained from citizen scientists and published literature

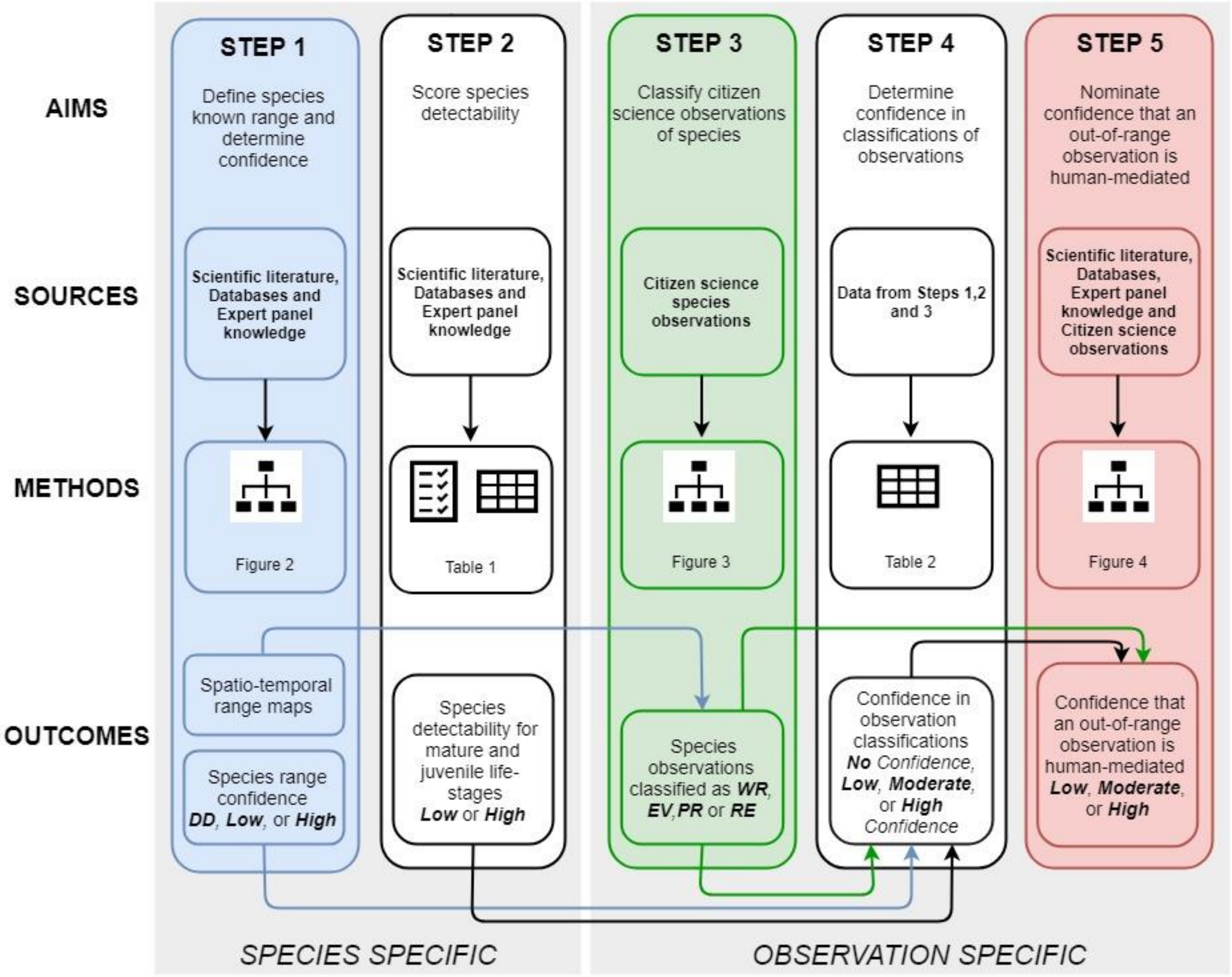


Figure 3.1 Overview of the approach to determine species' ranges, classify citizen science observations of species as out-of-range, and whether they are due to climate-mediated or human-mediated dispersal. Steps 1 through 5 use several data and knowledge sources and qualitative methods, including decision trees (e.g., Step 1), scoring and matrices (e.g., Step 2), to provide species-specific and observation-specific information (background grey boxes). The outcomes from each step cross-inform each other, and include rankings (e.g., 'LOW', 'HIGH') relevant to the aim of each step, or in the case of Step 3, a classification of the species occurrence is determined: within known range (WR), extralimital vagrancy (EV), potential range extension (PR), or range extension (RE). In Step 1, some species may be found data deficient (DD), and subsequent steps would not be undertaken.

are then classified as within known range, extralimital vagrancy, potential range extension, or range extension based on the observation information and their location relative to the defined spatio-temporal range (Step 3). To determine overall confidence in the classification of these observations, the species detectability score (from Step 2) and the measure of confidence in the species spatio-temporal range boundaries (from Step 1) are combined in Step 4. Last, observations that were classified as extralimital vagrancy, potential range extension, or range extension in Step 3 are further examined to assess whether they may be the result of human-mediated introduction in Step 5.

The decisions and scoring of species in Steps 1, 2, and 5 can be informed by knowledge of the target species taxonomy, appearance, geographic range, life-history traits and dispersal habits for example, provided by available scientific literature, databases (e.g., WoRMS, http://www.marinespecies.org/; GBIF, https://www.gbif.org; FishBase, https://www.fishbase.de/) and an expert panel (see 'Sources' in Figure 1). The formation of an 'expert panel' will be dependent on the study context and target species but could include professional scientists with specific taxonomic or regional expertise, citizen scientists, holders of traditional or local ecological knowledge, or natural historians (Johnson et al., 2017). The appropriate mode of expert knowledge elicitation will also vary according to the specific case-study but could include methods such as face-to-face meetings and facilitated discussions, or electronic questionnaires that are collated to provide majority opinions. The input of the expert panel is required for each target species (i.e., expert input is roughly scaled by the number of target species), whereas the classification of individual observations of target species can be conducted by a non-expert using the qualitative assessment tools.

Below we describe and demonstrate the stepwise approach using a case-study.

3.3.2 Suspected out-of-range marine fishes in Aotearoa New Zealand

For our case-study, New Zealand was defined as the entire coastline of the North and South Islands, and all offshore islands within the Exclusive Economic Zone (excluding the Rangitāhua/Kermadec Islands, Chatham Islands, and the Subantarctic Islands). We focused on marine teleost fishes that have been described in publications as either ‘invasive’, ‘alien’ ‘vagrant’, ‘occasional’, ‘tropical’, ‘subtropical’, ‘rare’ or ‘new-to-New Zealand’. Marine fishes have been identified as effective indicators of local climate change impacts in other locations (Holbrook, Schmitt, & Stephens, 1997; Last et al., 2011; Feary et al., 2013), but there has been no concerted effort to consolidate knowledge regarding distributions of fishes in New Zealand and there is no national monitoring network. In our case-study, Steps 1 and 2 were completed for 86 species to examine the method performance across species of varying biology, ecology, and levels of expert knowledge. Ten species that represented the extremes in the potential outcomes of Steps 1 and 2, and for which there were varying numbers of species occurrences, were then selected to complete Steps 3 to 5 providing a diverse test and demonstration of our method. (For further information on target species selection, see Appendix S2.)

Occurrence data for these ten target species were collated from peer-reviewed literature including citizen observations (e.g., Francis & Evans 1992; Francis, Worthington, Saul, & Clements, 1999; Roberts, Stewart, Struthers, Barker, & Kortet, 2017; Francis, 2019), unpublished occurrence records held by scientists, online databases (e.g., inaturalist, https://inaturalist.nz/), social media (e.g., Facebook and Instagram), online forums (e.g., New Zealand Fishing forum and New Zealand Spearfishing forum) and other opportunistic observations from citizens (see: Appendix S1 for all occurrence data). Occurrence data spanned 1975 to 2019 and were quality controlled prior to inclusion in the study (as described in Appendix S2).

An expert panel was formed to meet for a one-day workshop with subsequent email and verbal correspondence. Panel members consisted of New Zealand-based science professionals including: marine scientists (four individuals, with a combined total of 350+ peer-reviewed publications on species biology, marine biosecurity and population dynamics in New Zealand and the South Pacific), experienced taxonomists (two individuals with >15 years' experience each and >1200 combined citations) and resource managers (one individual with over 20 years' experience in fisheries regulation) representing eleven research institutions including universities, museums and government agencies. The panel also included experienced citizen scientists consisting of: spearfishers (three active individuals with recognised observational expertise and over 50 years' combined experience), fishers (two active individuals that represent both commercial and recreational sectors and engage through published media), and SCUBA divers (three experienced divers and underwater photographers with recognised observational expertise and over 50 years' combined experience).

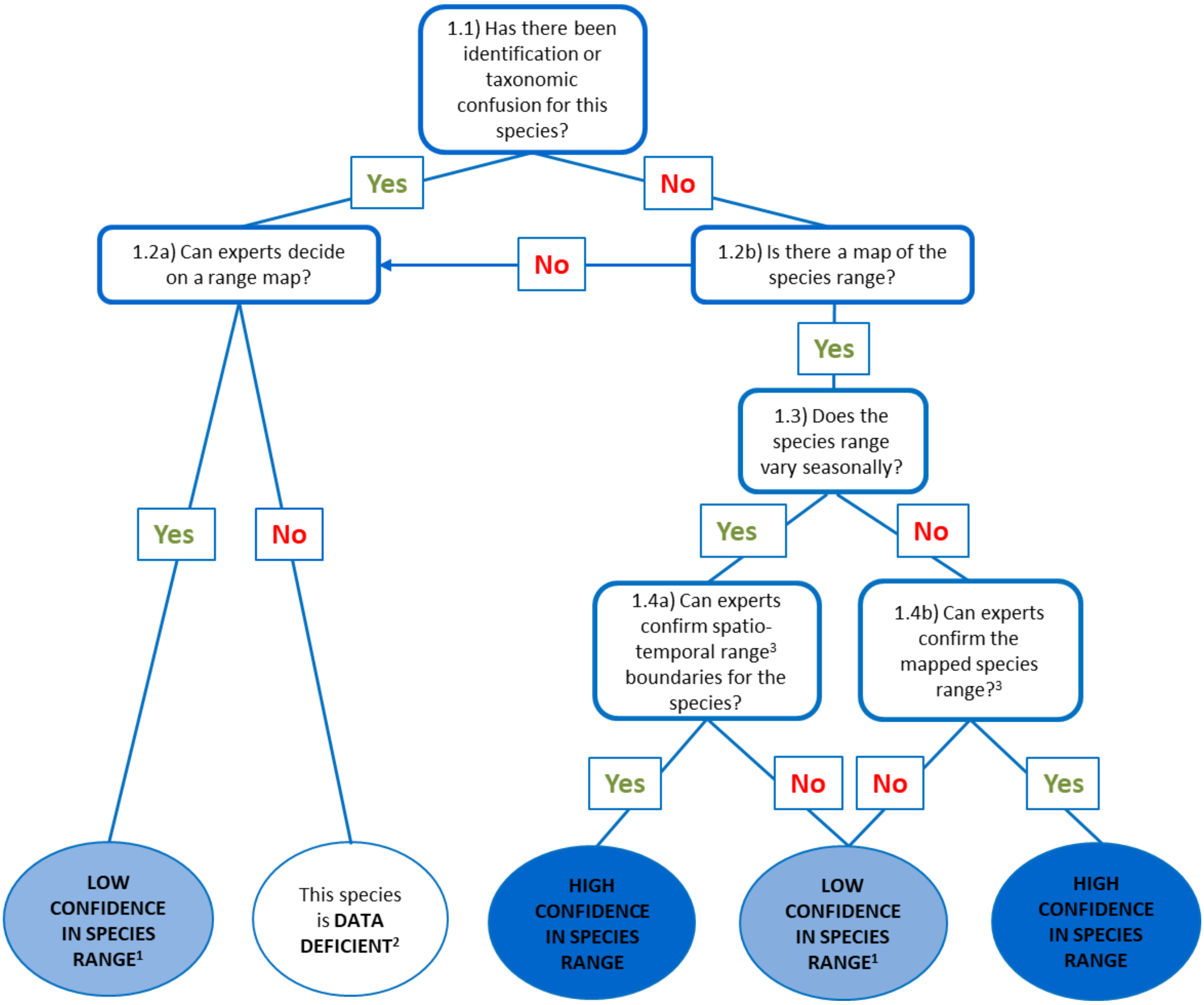
During the one-day workshop the panel was broken into three sub-groups and assigned a list of target species to work through Steps 1, 2 and 5. The sub-groups were organised based on expertise (i.e., Fishing, SCUBA, Spearfishing) and were assigned species according to the mode by which the species was typically observed. Several species were assigned across multiple sub-groups to determine if there were sub-group biases and to increase overall confidence. All step outcomes were reviewed by the entire panel and final decisions were made by group consensus. A detailed description of all the steps that follow and the evidence/knowledge-based decisions pertaining to the case-study are included in Appendix S2 and S4 respectively.

3.3.3 Step 1: Determining the species spatio-temporal range and expert confidence

Determining known ranges is essential to set a baseline for classifying out-of-range occurrences; however, experts will have varying confidence in the data underlying the proposed baselines (Bates et al., 2014; Robinson et al., 2015). In our method, we use a decision tree to help describe the spatio-temporal range (i.e., extent of the species' typical geographic distribution including any seasonal variation) with specific relevance to the case-study area and determine the collective confidence of the expert panel in the species’ range/s (Figure 2). In the case of pelagic or highly mobile species, distinct spatial ranges would be described for relevant times of the year (e.g., capturing migration for feeding or reproduction)

For each target species, the panel first answered Question 1.1; whether there was any taxonomic or identification confusion that could affect our ability to accurately define the species’ range. If a target species could be confused with a similar co-occurring species or there had been taxonomic confusion that may undermine knowledge of the species’ range, the panel answered ‘YES’ and moved to 1.2b. If the panel was confident there was no taxonomic confusion, they answered ‘NO’ to Question 1.1 and moved to 1.2a.

To assist the panel in addressing Question 1.2a, the authors constructed putative range maps for the target species in the study area using data sourced from the Ocean Biogeographic Information System (OBIS, www.iobis.org) and two New Zealand fish reference texts (Francis, 2012; Roberts et al., 2015). In assessing Question 1.2a, the panel verified and refined the putative range maps according to published resources and their own observations before moving to Question 1.3. If no mapped range was able to be sourced, the assessment moved to Question 1.2b. If the panel determined that the target species’ range from Question 1.2a varied seasonally (Question 1.3) they were prompted to describe the spatio-temporal range of this species in Question 1.4a. If the target species showed no seasonal variability (‘NO’ to Question 1.3) the assessment proceeded to Question 1.4b, where the accuracy of the species’ spatial range boundaries was determined. To summarise, the outcomes of Step 1 included a species’ spatio-temporal range map/s and an associated measure of confidence in that range.



[1] If the confidence in a species spatio-temporal range was 'LOW', obtaining more occurrence records could assist in increasing the confidence.

[2] If a species was classified as data deficient, further research should focus on morphology and genetic identification tools and/or species distribution surveys. [3] In Step 1.4 the expert panel is asked to confirm the current species mapped range based on an independent or corroborating secondary information source, such as quantitative survey data or unpublished knowledge.

Figure 3.2 Decision tree for determining the spatio-temporal range of a species (*i.e., extent of the species typical geographic distribution for a given season/s*) and the confidence of the expert panel in the mapped species' range

3.3.4 Step 2: Scoring species detectability

Detectability of a species is a function of its natural abundance, habitat preference, morphology, and behaviour (Robinson et al., 2015). Furthermore, the rate of encounter will be influenced by the species abundance in areas that are frequented by observers.

In the case of easy-to-detect species, we might assume that the observations, and thus our classifications, more accurately represent their realistic occurrences and range dynamics. Conversely, for difficult-to-detect species, we have lower confidence in the accuracy of our classifications because it is likely that there will be unobserved occurrences. Similar to Robinson et al., (2015), the panel scored species detectability as either ‘HIGH’ or ‘LOW’ according to their likely method of detection, abundance and conspicuousness (Table 1). The method of detection for the target species was classified as either ‘fishing’ (species predominantly sighted as caught fish by commercial or recreational fishers) or ‘underwater visual’ (species predominantly sighted in situ by divers, snorkelers, or spearfishers) by the authors before the workshop and verified by the panel. Abundance was based on both the likelihood of encounter by observers (i.e., for underwater visual, how common the target species habitat is within diving range [rare = 1, occasional = 2, or common = 3]; or for fishing, how often a habitat is targeted by fishers [rare = 1, occasional = 2, or common = 3]) and the abundance of the target species within this habitat in its known range [rare = 1, patchy = 2, common = 3]). If the combined score was five or over a species was considered ‘abundant’, if the score was less than five a species was considered ‘not abundant’ (Table 1). Because conspicuousness, and the ability for citizen scientists to observe individuals in situ (i.e., underwater visual) can vary significantly with ontogeny we scored the conspicuousness of mature and juvenile life-stages separately. To assess conspicuousness, a species was scored for the presence of each of the following physical and behavioural characteristics: maximum size 30cm or larger = 1, does not camouflage = 1, and does not hide = 1. A species (or life-stage) was deemed ‘conspicuous’ if the total score was greater than or equal to two; if the score was less than two it was deemed ‘inconspicuous’ (Table 1). Abundance and conspicuousness were considered equal in importance when scoring detectability of underwater visual species, if either was ‘HIGH’ detectability was also ‘HIGH’. However, for fishing species, only abundance was considered as species are not observed in situ; if abundance was ‘HIGH’ detectability was also ‘HIGH’ (Table 1).

Table 3.1 Detectability scoring method (Step 2). (a) Abundance scoring for species observed predominantly by *fishing* and *underwater visual* observation, and (b) Conspicuousness scoring for species detected using *underwater visual* observation. A species (or life-stage) was deemed 'conspicuous' if the total score was greater than or equal to two; if the score was less than two it was deemed 'inconspicuous'. The detection method, abundance and conspicuousness scores were combined to inform an overall level of 'detectability' (c) for both mature and juvenile life stages of target species.

(a) Species abundance	**Score**
How commonly is the species habitat observed[1] or fished?[2]	
Rarely	1
Sometimes	2
Commonly	3
How common is the species in its habitat?[3]	
Rare	1
Patchy	2
Common	3

(b) Species conspicuousness	**Score**
Camouflage[4]	
Species exhibits some form of camouflage	0
Species does not exhibit camouflage	1
Body Size[5]	
Maximum adult body size smaller than 30cm	0
Maximum adult body size 30cm or larger	1
Hiding[6]	
Species hides behind structures or within substrate	0
Species does not hide	1

(c) Detectability scoring

Detection method	*Abundance (a)*	*Conspicuousness (b)*	*Detectability*
Underwater visual	Abundant	Conspicuous	HIGH
Underwater visual	Abundant	Inconspicuous	HIGH
Underwater visual	Not abundant	Conspicuous	HIGH
Underwater visual	Not abundant	Inconspicuous	LOW
Fishing	Abundant	NA	HIGH
Fishing	Not abundant	NA	LOW

[1] in areas that observers in New Zealand may dive, spearfish, or swim.
[2] including recreational and commercial fishing in New Zealand waters.
[3] within the species' known range.
[4] camouflage includes background matching, self-shadow concealment, obliterative shading (countershading), and masquerade (leaf/other fish).
[5] body size was obtained from peer-reviewed literature, online databases, or expert knowledge.
[6] if a species is known to hide, i.e., it generally conceals itself behind an object or inside cracks and crevices. Behavioural data were obtained from peer-reviewed literature or expert knowledge.

3.3.5 Step 3: Classifying citizen science observations of species

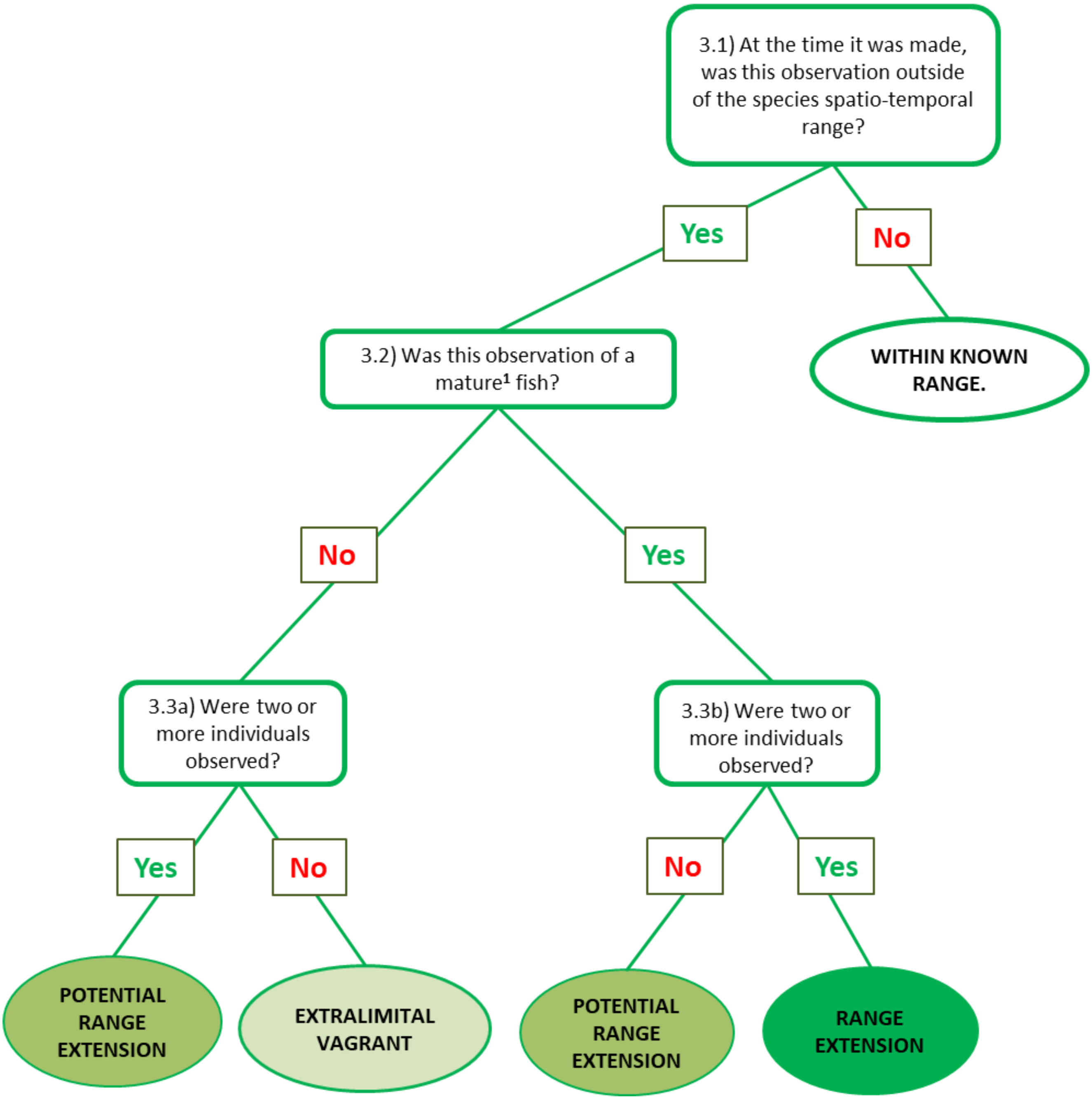
A decision tree (Figure 3) was used to classify observations of target species as one of the following: within known range, extralimital vagrancy, potential range extension or range extension. By assessing each observation separately, we acknowledge that different parts of a species' range may be undergoing different range dynamics, and that range extensions and species invasions can be stepwise (Bates et al., 2014). Thus, separate occurrences for the same target species may be classified differently but can be aggregated to form an overall view of the species range dynamics. The species' spatio-temporal range determined by the panel from Step 1 was used to determine if the species observation was out-of-range (Question 3.1). If the observation was out-of-range, the species maturity was assessed (Question 3.2). Features of maturity vary among species but could include being of, or near, maximum size and/or reproductive size, mature colouration, or spawning/nesting behaviour. In Question 3.3 the number of individuals observed was assessed. If mature individuals or two or more individuals were sighted during the same observation event, there is a higher probability of the species being able to reproduce and/or persist, and thus, represent a range extension. These three questions (Questions 3.1, 3.2, and 3.3) were used to distinguish an out-of-range occurrence as extralimital vagrancy, potential range extension, or range extension.

Table 3.2 Confidence scoring method for the classification of observations (Step 4). Overall confidence scores are a function of the spatio-temporal range confidence (Step 1) and detectability (Step 2). Range confidence can vary seasonally (i.e., a summer range and a winter range) and/or across life-stages (i.e., juvenile, and mature).

		Step 1: Spatio-temporal range confidence	
		LOW	*HIGH*
Step 2: Species detectability	*LOW*	**NO** Confidence	**MODERATE** Confidence
	HIGH	**LOW** Confidence	**HIGH** Confidence

3.3.6 Step 4: Determining confidence in the classification of observations

In some applications, a score of overall confidence in the classification of species occurrences may be helpful. Increased confidence in a species' spatio-temporal range gives greater confidence in the identification of out-of-range occurrences. However, a species' detectability also contributes to our overall confidence because it influences the likelihood that our knowledge based on observations accurately represents the species range and actual occurrence within a location. Thus, in Step 4, we combine species' range confidence (Step 1) and species detectability scores (Step 2) to provide an overall confidence in the classification of observations for that species (Table 2).

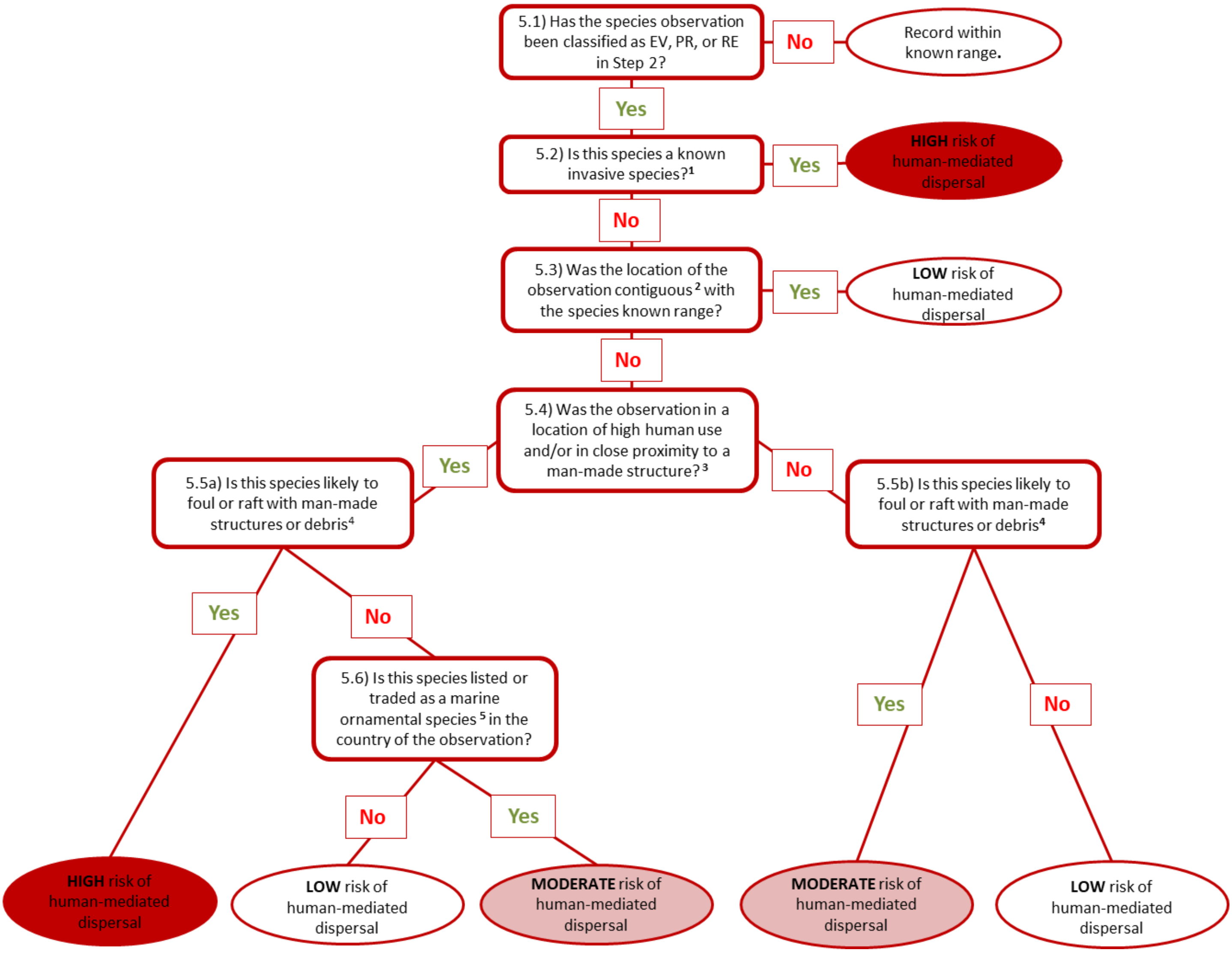


[1] The features used to determine maturity in Step 3.2 may vary among species but are likely to include reaching maximum size and/or reproductive size, mature colouration or spawning/nesting behaviour

Figure 3.3 Decision tree for classifying species observations as within known range, extralimital vagrancy, potential range extension, or range extension using the spatio-temporal range determined by the expert panel in Step 1

3.3.7 Step 5: Determining confidence that out-of-range occurrences are due to human-mediated dispersal

Several potential pathways for human-mediated dispersal of marine species have been identified globally, including hull fouling, ballast discharge, the aquarium trade, aquaculture, and intentional introductions (Lonhart, 2009; Rilov & Crooks, 2009). For out-of-range occurrences from Step 3, the panel assigned a level of confidence (‘LOW’, ‘MODERATE’ or ‘HIGH’) as to whether the occurrence may be a result of human-mediated introduction (Figure 4). To do so, we assessed if the target species was known to, or likely to, be spread by humans (e.g., a known introduced alien species listed in online or published databases; Question 5.2) or if it possessed traits known to heighten the risk of human-mediated introduction, such as rafting with debris or associating with biofouling on vessel hulls (Question 5.5a,b). We also assessed if the species’ known range was contiguous with the out-of-range occurrence, where the assumption was that human-mediated introductions lead to discontiguous species’ ranges (Question 5.3) centred around areas of anthropogenic activity such as ports, wharves, or popular anchorages (assessed in Question 5.4; as per Chapman & Carlton, 1991; Sorte et al., 2010). These are likely areas of higher vessel movement, which in turn increase the likelihood of species introductions by way of hull fouling, bilge, or ballast water discharge. If species were found away from areas of high human use, the propensity of a species to raft with, or foul, man-made structures or debris was still assessed (Question 5.5b). Such vessel and debris movements are known to have introduced marine species to areas remote from centres of anthropogenic activity (Francis et al., 1999; Willis, Saunders, Blackwood, & Archer, 1999; Wotton & Hewitt, 2004). However, we only consider the release of ornamental or aquarium species (Question 5.6) if they occur in areas of high human use (i.e., we answered ‘YES’ to Question 5.4).



[1] Classified as an unwanted organism or pest by a regional authority (e.g., Biosecurity New Zealand), or listed on an international unwanted species register such as the Global Invasive Species Database or the National Exotic Marine and Estuarine Species Information System (NEMESIS) database. [2] A contiguous range is continuous spatially and not patchy or geographically disjunct. [3] Such as wharves, marinas, marine farms, mooring areas and harbours in, and adjacent to, urban areas. [4] Such as floating plastic, fishing debris, macroalgal rafts, terrestrial plant material, vessels and marine structures. [5] Such as the New Zealand Ministry for Primary Industries “Schedule 4 Approved species of ornamental fish and marine invertebrates” (document ORMAMARI.ALL, 13 April 2017).

Figure 3.4 Decision tree for determining confidence in out-of-range observations classified in Step 3 (extralimital vagrant, potential range extension, or range extension) as human-mediated\

3.4 Results

A total of 86 species were scored for range confidence (Step 1) and detectability (Step 2) by the expert panel during the one-day workshop. Thirty-three of these species were independently scored by more than one sub-group to address observational and expert bias before the panel made a consensus decision. For Step 1, the panel could not determine a spatio-temporal range or could not reach consensus for an appropriate spatio-temporal range, for 25 species (29%), deeming them Data Deficient. For 13 (52%) of these species, identification, or taxonomic confusion (Question 1.1 in Figure 2) contributed to their data deficiency, and for 12 species (48%) there was no range map available (Question 1.2b). Most of the Data Deficient species (80.2%) were predominantly observed in situ (i.e., underwater visual species). These 24 Data Deficient species were excluded from further assessment steps. For the remaining 62 species, the expert panel defined spatio-temporal ranges with 'HIGH' confidence for 35 species (56%) and 'LOW' confidence for 27 species (44%), with 16 species (27%) having species ranges that varied seasonally (Question 1.3). Most species for which there was 'HIGH' confidence in the spatio-temporal ranges were underwater visual species (95.2%; see Appendix S3 for supplementary results).

In Step 2, the panel scored 24 underwater visual species as 'HIGH' for detectability for both the juvenile and mature stages, 9 species as 'LOW' for detectability for both the juvenile and mature stages, and 15 with differing detectability scores across life-stages (Table 3). For fishing species, the panel scored 5 species as 'HIGH' for detectability for both the juvenile and mature stages, 6 species as 'LOW' for detectability for both the juvenile and mature stages, and 4 with differing detectability scores across life-stages.

Before declaring a consensus, the most notable differences in the sub-group outcomes for Step 2 were the scores of the Fishers sub-group compared to the other two sub-groups (SCUBA and Spearfishing; Table 3). The Fishers sub-group scored a larger proportion of the juvenile life-stages as ‘conspicuous’ and consequently, the detectability was also ‘HIGH’, leaving no difference between the scored detectability of juvenile and mature life-stages. These differences in initial sub-group scoring (and previously for ranges in Step 1) were resolved through panel discussion and consensus before proceeding to Step 3.

The ten species selected for Step 3 spanned a broad range of habitats, life history, behavioural traits (see Appendix S2), likelihood of encounter, as well as varying levels of prior knowledge regarding the species in New Zealand.

Table 3.3 Summary of the proportion of juvenile and mature life-stages classified as abundant (i.e., 'HIGH' abundance), conspicuous (i.e., 'HIGH' conspicuousness) and highly detectability for target species assessed by more than one sub-group of the expert panel.

Factor	**Panel sub-group**	**Total number of species assessed (n)**[1]	**Juvenile**		**Mature**	
			LOW[2]	*HIGH*[2]	*LOW*[2]	*HIGH*[2]
Abundance	*Fishers*	15	0.93	0.07	0.93	0.07
	SCUBA divers	13	0.77	0.23	0.69	0.31
	Spearfishers	27	0.96	0.04	0.81	0.19
Conspicuousness	*Fishers*	9	0.11	0.89	0.11	0.89
	SCUBA divers	13	0.54	0.46	0	1
	Spearfishers	21	0.52	0.48	0	1
Detectability	*Fishers*	9	0.11	0.89	0.11	0.89
	SCUBA divers	13	0.46	0.54	0	1
	Spearfishers	21	0.59	0.41	0	1

[1] Both 'fishing' and 'underwater visual' species were assessed for abundance; only 'underwater visual' (species that are observed *in situ*) were assessed for conspicuousness. [2] Abundance was 'HIGH' if the score was ≥ 5; Conspicuousness was 'HIGH' if the score was ≥ 2; Detectability was based on the method of detection, abundance and conspicuousness.

In the case of two species, Tailor (*Pomatomus saltatrix*, Linnaeus, 1766), and Wahoo (*Acanthocymbium solandri*, Cuvier, 1832) the panel could not confidently define their ranges in Step 1 (described above), so despite having citizen observations of the species, they could not be classified in Step 3 (Table 4). In the case of Tailor, an insufficient number of observed species occurrences prevented the panel from establishing the species' range in New Zealand. Although there were occurrence records for Wahoo, the high mobility of this species and mistrust of some of the data meant a consensus could not be reached regarding the range boundaries for this species in New Zealand.

The citizen science observations for the remaining eight target species were classified by the authors as within known range, extralimital vagrancy, potential range extension or range extension (Step 3; Figure 3). To provide a measure of confidence in these classifications, the panel's outcomes for Steps 1 and 2 were combined in Step 4 (Table 2). For two species, Painted moki (*Cheilodactylus ephippium*, McCulloch & Waite, 1916) and Spotted-black grouper (*Epinephelus daemelii*, Günther, 1876), all citizen science observations were within the species' known range (Figure 5). For Painted moki in particular, we had 'HIGH' confidence in the defined range (Table 4), providing no evidence that the species' range is changing in New Zealand. We identified four species – Eye-stripe surgeonfish (*Acanthurus dussumieri*, Valenciennes, 1835), Lord Howe coralfish (*Amphichaetodon howensis*, Waite, 1903), Mahimahi (*Coryphaena hippurus*, Linnaeus, 1758), and Horned blenny (*Parablennius intermedius*, Ogilby, 1915) – that had occurrences classified as range extensions (Figure 5). For Lord Howe coralfish and Mahimahi, we had 'HIGH' confidence that these occurrences represented true range extensions. Southern demoiselle (*Chrysiptera notialis*, Allen, 1975), Mahimahi and Sergeant major (*Abudefduf vaigiensis*, Quoy & Gaimard, 1825) also had observations that were classified as potential range extensions (Figure 5). We had the highest overall confidence in the out-of-range observations for Mahimahi and mature Southern demoiselle (Table 4; see Appendix S4 for supplementary results).

Finally, in Step 5 the panel assessed whether any out-of-range species occurrences were likely the result of human-mediated introduction (Step 5; summarised in Table 4). All decisions in Step 5 were made using group consensus, but there was no disagreement in outcomes among panel members. For the six species that had out-of-range occurrences, all occurrences were 'LOW' risk of being facilitated by human-mediated introduction, except the sole occurrence of the Horned blenny (Table 4).

TABLE 3.4 Summary of results for the ten target species, including confidence in spatio-temporal range boundaries (Step 1), detectability (Step 2) and overall confidence in the observation classifications (Step 4). The proportion of the species observations classified within known range (WR), extralimital vagrancy (EV), potential range extensions (PR) and range extensions (RE) from Step 3, and the probability that the EV, PR, or RE observations were facilitated by human-mediated dispersal (Step 5). Species that were data deficient or had no out-of-range observations are denoted by NA. Values or descriptions in brackets indicate results for juvenile stages where they differed from mature stages. For full step-by-step results see Appendix S2 in Supporting Information.

Species	**Common name**	**Step 1: Range confidence**	**Step 2: Detectability**		**Step 4: Overall confidence**	**Number of citizen science observations**	**Step 3: Classification of observations (as proportions of total observations)**				**Step 5: Confidence in human-mediated dispersal (as proportions of total observations)**		
			Juvenile	**Mature**			**WR**	**EV**	**PR**	**RE**	**LOW**	**MOD**	**HIGH**
Abudefduf vaigiensis	Sergeant major	LOW	HIGH	HIGH	LOW	9	0.75	0	0.25	0	1	0	0
Acanthocybium solandri	Wahoo	DD	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Acanthurus dussumieri	Eye-stripe surgeonfish	LOW	LOW	HIGH	LOW (NO confidence)	20	0.93	0	0	0.07	1	0	0
Amphichaetodon howensis	Lord Howe coralfish	HIGH	HIGH	HIGH	HIGH	124	0.98	0	0	0.02	1	0	0
Cheilodactylus ephippium	Painted moki	HIGH	HIGH	HIGH	HIGH	34	1	0	0	0	NA	NA	NA
Chrysiptera notialis	Southern demoiselle	HIGH	LOW	LOW	LOW	3	0	0.33	0.67	0	1	0	0
Coryphaena hippurus	Mahimahi	HIGH	HIGH	HIGH	HIGH	59	0.94	0	0.02	0.02	1	0	0
Epinephelus daemelii	Spotted-black grouper	LOW	LOW	HIGH	LOW (NO confidence)	28	1	0	0	0	NA	NA	NA
Parablennius intermedius	Horned blenny	HIGH	LOW	LOW	MOD	1	0	0	0	1	0	0	1
Pomatomus saltatrix	Tailor	DD	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA

Although we had ‘MODERATE’ confidence that the observation of the Horned blenny represented a true range extension (Figure 5), we had ‘HIGH’ confidence that this out-of-range occurrence was facilitated by human-mediated introduction (Step 5). This outcome was based on the observation of the species on a man-made structure, and in a location that was discontiguous with the known species range.

3.5 Discussion

Our approach draws on the strengths of expert knowledge and opportunistic citizen science observations to infer the range dynamics of previously unmonitored species. Here, we provide proof-of-concept for the method by classifying the range dynamics of several marine fishes in Aotearoa New Zealand. The ease of our approach and the intuitive outputs it produces should appeal to managers and science practitioners concerned with climate-induced biodiversity changes and the detection of alien species. Furthermore, the approach is modular and flexible, can be varied to enable application across different taxonomic groups and ecosystems and in combination with other existing tools. Below we highlight the novel features of our approach exemplified by our case-study, and discuss the methods performance based on the diversity of target species we used, the extent of knowledge and data regarding these target species, as well as the expert panel formation and mode of elicitation.

Climate-mediated range extensions are often diffuse and difficult to differentiate from human-mediated species introductions (Sorte et al., 2010). Nonetheless, this distinction may be important for resource managers looking to monitor climate change impacts and/or detect dispersal pathways of alien species. For instance, New Zealand is heavily reliant on maritime transport of commodities and a popular destination for overseas recreational vessels, creating a significant pathway for marine species introductions through biofouling and ballast water (Coutts & Taylor, 2004; Dodgshun, Taylor, & Forrest, 2007), yet novel species introductions are also known to occur via oceanic dispersal (Francis & Evans, 1992), and these are likely to increase in future (Molinos et al., 2016). The final step in our method (Step 5) helps discriminate between human-mediated introductions and climate-mediated range extensions. In our case-study, we were able to determine that an observation of multiple mature individuals of the Horned blenny (*P. intermedius*) on a man-made wharf in north-eastern New Zealand was likely to have been facilitated by human-mediated vectors (Figure 5). The detection of this species, and other human-mediated species introductions, help to inform the appropriate practical responses of managers to out-of-range species occurrences (Bates et al., 2014; Weir & Salice, 2011).

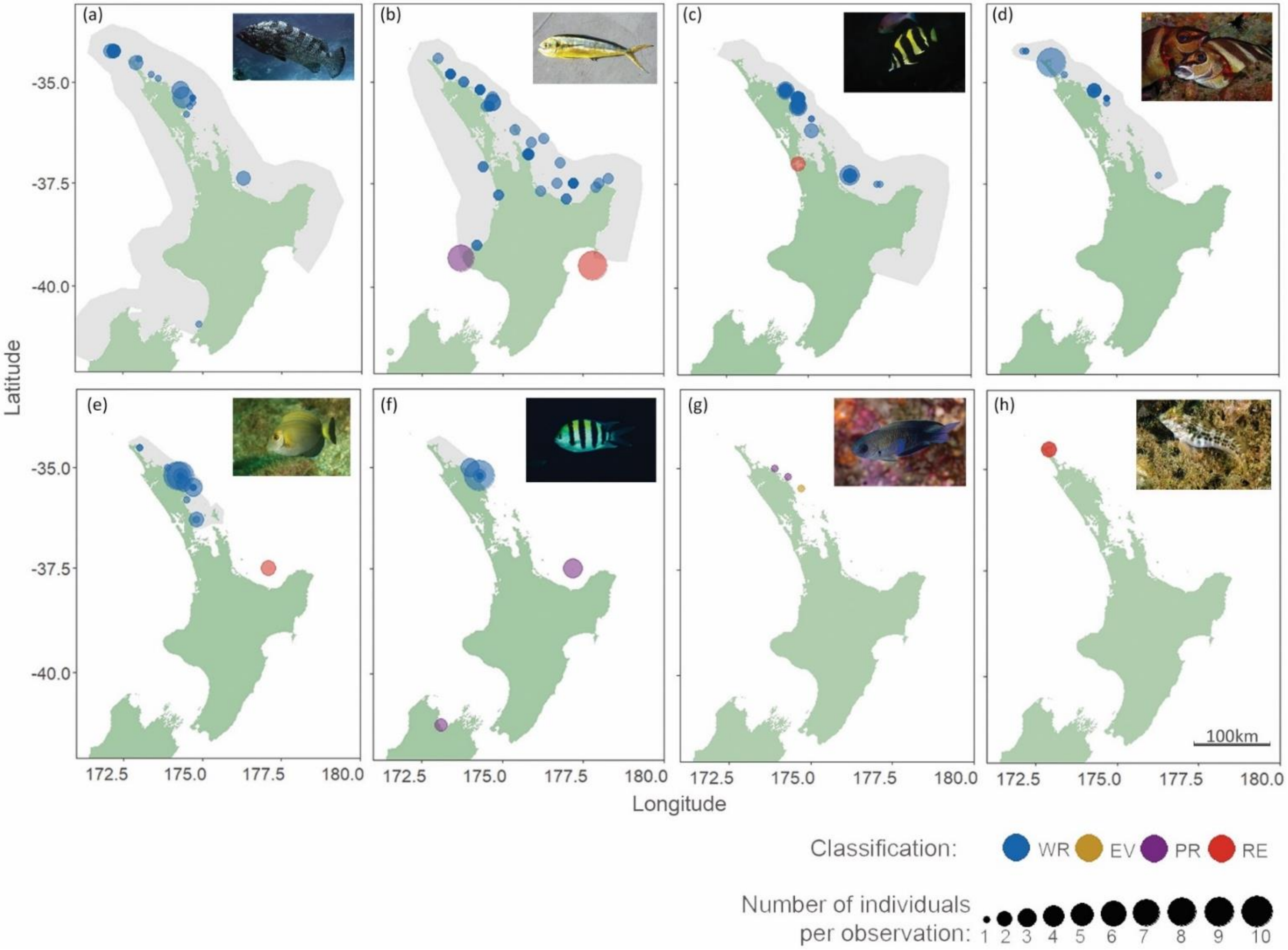


FIGURE 3.5 Observation classifications (within known range, WR; extralimital vagrant, EV; potential range extension, PR; range extension, RE; Step 3) for the target species where experts were able to define spatio-temporal ranges (grey shading; Step 1). (a) Spotted-black grouper, *Epinephelus daemelii*; (b) Mahimahi, *Coryphaena hippurus*: the species' range during the New Zealand summer season is shown, during the winter the species is largely absent from New Zealand waters; (c) Lord Howe coralfish, *Amphichaetodon howensis*; (d) Painted moki, *Cheilodactylus ephippium*; (e) Eye-stripe surgeonfish, *Acanthurus dussumieri*; (f) Sergeant major, *Abudefduf vaigiensis*; (g) Southern demoiselle, *Chrysiptera notialis*; and (h) Horned blenny, *Parablennius intermedius*: the expert panel determined that the species spatio-temporal range did not include New Zealand, and were confident that the species occurrence (classified as RE) was due to human-mediated dispersal (established in Step 5).

Identifying specific types of change in species range dynamics, such as increased arrival, greater survival to maturity and seasonal changes in species' ranges (i.e., phenological changes), is critical in anticipating future biodiversity impacts. By including contextual information regarding the observation (i.e., season, maturity, number of individuals), our approach allows for the differentiation between the arrival (i.e., extralimital vagrancy), population growth and persistence stages (i.e., potential range extensions and range extensions) of introduced aliens or range extenders (*sensu* Bates et al., 2014). For instance, the pelagic Mahimahi (*C. hippurus*) is highly mobile and has distinct seasonal ranges in high latitude locations (Norton, 1999). Sporadic occurrences of juvenile Mahimahi in New Zealand in summer are likely to be vagrancies, but our detection of several mature fish sighted outside of their spatio-temporal range, represents range extensions (Figure 5). Through the precise use of observation information and the classification of individual occurrences, our method provides a means to continuously monitor species range dynamics, helping to detect real-time, spatially explicit range changes that aid in understanding the physical and environmental drivers of range dynamics.

Areas with a high incidence of marine range extensions have experienced devastating biodiversity and ecosystem changes (Hawkins et al., 2009; Ling, 2008; Pecl et al., 2017; Wernberg et al., 2011). Our method can help to highlight geographic areas where there is a high number of range extending (PR or RE) species occurrences (Figure 5), and similarly to Lonhart, Jeppesen, Beas-Luna, Crooks, & Lorda (2019), could be used to identify spatial hotspots for range extensions across several species. Most of the range extensions identified in our case-study occurred within the Northeastern and Portland bioregions of New Zealand (Shears, Smith, Babcock, Duffy, & Villouta, 2008); regions where rare and vagrant species have previously been observed (Francis, 1996; Francis et al., 1999; Roberts et al., 2015). Continued monitoring could be prioritised in such regions to allow for the timely identification of new range extensions and their potential impacts, and to inform management interventions.

The use of citizen science observations offers a cost-effective and growing opportunity to increase the scope and resolution of spatial and temporal data to monitor species range dynamics (Devictor, Whittaker, & Beltrame, 2010). In our case-study, all ten target species occurrences classified as potential range extensions or range extensions (Table 4) were made by citizen scientists and were previously unknown to the expert panel. Consequently, the power of the approach may be in part determined by the spatial and temporal patterns of citizen science observations (Bird et al., 2014). Although our approach cannot address range dynamics where there are no citizen science observations, the use of expert knowledge and published literature to cross-inform classifications, reduces the risk of inferring out-of-range occurrences and changes in range dynamics erroneously where there are observations. For instance, in our case-study, the proportion of out-of-range occurrence classifications was unrelated to the number of observations for the target species (Table 4).

The classification of species range dynamics, and confidence in those classifications, hinges on expert knowledge regarding: the spatio-temporal ranges of target species (Step 1), their perceived detectability (Step 2), and propensity to be dispersed by humans (Step 5). For our case-study, we formed an expert panel that maximised the breadth of knowledge across New Zealand and South Pacific fishes, as well as modes and geographic regions of encounter with these species. Owing to the diverse knowledge bases of the expert panel, frequently sub-groups arrived at different outcomes for certain steps of the approach (Table 3). It is common practice to convene a panel of experts and ask them to produce consensus decisions (Johnson et al., 2017). In our case-study, panel members were able to self-identify their weaknesses and support the evidence presented by other panel members to reach consensus decisions in most cases. Despite the diversity of expertise and willingness to reach an amicable consensus, our approach could not be applied to 29% of the target species because of a lack of knowledge and/or access to appropriate data (classified as Data Deficient). In this way, our approach has helped identify knowledge gaps that limit our ability to monitor the range dynamics of these species, helping to prioritise future research such as targeted taxonomic work or survey of underrepresented regions.

Using our approach, we were able to classify the range dynamics for a diversity of fishes, including benthic, pelagic, and fished species, including those that varied in detectability. For our ten target species, their detectability did not affect the ability of the expert panel to determine their spatio-temporal ranges with high confidence; rather, the expert panel had 'LOW' confidence in the spatio-temporal ranges of species with 'detectability' scored 'HIGH' (with the exception of juvenile Spotted-black grouper, *E. daemili* and Eyestripe surgeonfish, *A. dussumieri*; Table 4). The classification of species occurrences was also not evidently influenced by the detectability of a species. For example, occurrences for two species that had 'LOW' detectability (Southern demoiselle, *C. notialis* and Horned blenny, P. intermedius) were classified as potential range extensions (Table 4). Pelagic, migratory species were frequently data deficient due to a lack of information regarding their spatio-temporal ranges (e.g., Tailor and Wahoo), and based on discussion among expert panel members, co-occurring congenerics and conspecifics (e.g., Kyphosidae) are also likely to be data deficient using our approach owing to a lack of distinguishing features leading to taxonomic confusion.

The life-history traits and questions included in our method are ubiquitous indicators for the ability of a species to extend its range in both marine and terrestrial systems. Furthermore, the steps of our approach may be easily machine-automated and refined to suit other data sources, geographic settings, or complemented with new steps and existing tools (e.g., the Marine Screening-Level Risk Assessment Protocol for Marine Non-Indigenous Species [DFO, 2015]; Marine Fish Invasiveness Scoring Kit [MFISK], www.cefas.co.uk). For instance, the management response to an out-of-range occurrence is likely to be determined by the stage of establishment, species traits and the potential impacts of the alien species, regardless of whether its introduction has been climate-mediated or human-mediated. Such risk or impact assessments could be additional steps in the method, triggered when an out-of-range occurrence is indicated by our method. The value and utility of such a modular and stepwise approach may be particularly high when there are minimal resources available for continuous monitoring, and the expertise and information required must draw on several specialist or management groups and include stakeholder participation.

3.6 Authors' Contributions

IM and LL conceived and IM, LL, JDA and TT designed the methodology and research. IM, collected data from citizen scientists, developed quality control measures and collated expert knowledge. TT, MF and CD contributed data and expert knowledge for the development of species range maps and for the case study and contributed to the expert determined steps in the methodology. IM and JDA analysed the data and produced the figures. IM drafted the initial manuscript and all authors contributed towards the interpretations of results, the initial drafting and the revisions of the final manuscript.

3.7 Supporting information

This chapter has been adapted from the following published paper:

Middleton, I., Aguirre, J. D., Trnski, T., Francis, M., Duffy, C., & Liggins, L. (2021). Introduced alien, range extension or just visiting? Combining citizen science observations and expert knowledge to classify range dynamics of marine fishes. Diversity and Distributions, n/a(n/a) https://doi.org/10.1111/ddi.13273

3.8 Appendix

3.8.1 Appendix 1: Supplementary occurrence records for ten target species from the case-study

Table S1.1 Citizen science occurrence data for ten target species including metadata that accompanied each observation. Classifications were determined by the authors using the decision tree in Step 3 of the assessment.

Species	**Number**	**Life stage**	**Year**	**Month**	**Latitude**	**Longitude**	**Classification**	**Location**
Abudefduf vaigiensis	1	Juvenile	2015	3	-35.225037	174.270947	WR	Bay of Islands
Abudefduf vaigiensis	1	Juvenile	2015	2	-35.191872	174.292573	WR	Bay of Islands
Abudefduf vaigiensis	1	Juvenile	2016	2	-35.191872	174.292573	WR	Bay of Islands
Abudefduf vaigiensis	5	Juvenile	2016	2	-34.997889	173.964701	WR	Far North
Abudefduf vaigiensis	1		2016	2	-35.171298	174.330482	WR	Bay of Islands
Abudefduf vaigiensis	20	Mature	2016	2	-35.225037	174.270947	WR	Bay of Islands
Abudefduf vaigiensis	2	Mature	2016	1	-35.183934	174.303562	WR	Bay of Islands
Abudefduf vaigiensis	6	Juvenile	2016	1	-37.522098	177.171398	PR	Bay of Plenty
Abudefduf vaigiensis	2	Sub Adult	2016	1	-41.210451	173.083833	PR	Nelson
Acanthocybium solandri	1	Unknown	2016	1	-35.150783	174.146366	N/A	Bay of Islands
Acanthocybium solandri	1	Unknown	2016	1	-35.619373	174.547785	N/A	Tutukaka
Acanthocybium solandri	1	Unknown	2016	4	-36.53	-174.43	N/A	(NMNZ P35975)
Acanthocybium solandri	1	Unknown	2017	2	-37.688006	176.534593	N/A	Bay of Plenty
Acanthocybium solandri	1	Unknown	2017	4	-35.171298	174.330482	N/A	Bay of Islands
Acanthocybium solandri	1	Unknown	2018	2	-35.47128	174.731295	N/A	Poor Knights Islands
Acanthocybium solandri	2	Unknown	2018	2	-35.244164	174.211652	N/A	Bay of Islands
Acanthocybium solandri	3	Unknown	2018	1	-34.415971	173.048143	N/A	Far North
Acanthocybium solandri	2	Unknown	2018	2	-34.415971	173.048143	N/A	Far North

Acanthocybium solandri	1	Unknown	2018	2	-35.150783	174.146366	N/A	Bay of Islands
Acanthocybium solandri	1	Unknown	2018	2	-35.47128	174.731295	N/A	Poor Knights Islands
Acanthocybium solandri	1	Unknown	2019	2	-35.47128	174.731295	N/A	Poor Knights Islands
Acanthocybium solandri	1	Mature	2019	2	-36.964628	176.081154	N/A	Coromandel
Acanthocybium solandri	1	Mature	2019	3	-35.043371	173.738477	N/A	Far North
Acanthocybium solandri	1	Mature	2019	3	-37.651999	176.165167	N/A	Bay of Plenty
Acanthocybium solandri	1	Mature	2019	1	-34.826843	173.157514	N/A	Far North
Acanthocybium solandri	1	Sub Adult	2019	1	-35.3895	174.794667	N/A	Poor Knights Islands
Acanthocybium solandri	1	Sub Adult	2019	2	-34.826843	173.157514	N/A	Far North
Acanthocybium solandri	1	Sub Adult	2019	1	-35.200302	174.269085	N/A	Bay of Islands
Acanthurus dussumieri	2	Juvenile	2001	4	-35.225037	174.270947	WR	Bay of Islands
Acanthurus dussumieri	8	Juvenile	2018	11	-35.225037	174.270947	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	1986	12	-35.220382	174.289214	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	1994	4	-35.225037	174.270947	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	1978	6	-35.481201	174.736785	WR	Poor Knights Islands
Acanthurus dussumieri	1	Juvenile	2019	2	-34.519629	172.964103	WR	Far North
Acanthurus dussumieri	1	Juvenile	2019	3	-35.171298	174.330482	WR	Bay of Islands
Acanthurus dussumieri	2	Juvenile	2011	5	-36.288167	174.81954	WR	Hauraki Gulf
Acanthurus dussumieri	8	Juvenile	2014	1	-35.223314	174.19055	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	2011	3	-36.288167	174.81954	WR	Hauraki Gulf
Acanthurus dussumieri	2	Mature	2011	3	-37.477077	177.133173	RE	Bay of Plenty
Acanthurus dussumieri	1	Juvenile	2015	5	-35.829489	174.4983	WR	Whangarei
Acanthurus dussumieri	1	Juvenile	2010	4	-35.453259	174.733188	WR	Poor Knights Islands
Acanthurus dussumieri	3	Juvenile	1993	2	-35.479909	174.736158	WR	Poor Knights Islands
Acanthurus dussumieri	1	Juvenile	1993	2	-35.225037	174.270947	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	1993	2	-35.03421	173.918911	WR	Far North

Acanthurus dussumieri	2	Juvenile	1993	2	-35.437907	174.40639	WR	Far North
Acanthurus dussumieri	1	Sub Adult	1993	2	-35.225037	174.270947	WR	Bay of Islands
Acanthurus dussumieri	1	Juvenile	1993	2	-34.519629	172.964103	WR	Far North
Acanthurus dussumieri	1	Juvenile	1993	2	-35.481201	174.736785	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	1998	2	-35.941766	175.14791	WR	Auckland
Amphichaetodon howensis	1	Unknown	1998	3	-35.461825	174.741654	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	1998	9	-35.941766	175.14791	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2017	3	-35.200302	174.269085	WR	Bay of Plenty
Amphichaetodon howensis	1	Mature	2016	4	-35.191872	174.292573	WR	Mokohinau Islands
Amphichaetodon howensis	1	Mature	2017	4	-35.200302	174.269085	WR	Mokohinau Islands
Amphichaetodon howensis	1	Mature	2017	8	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2016	12	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	2	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2015	1	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	1	-35.200302	174.269085	WR	Bay of Plenty
Amphichaetodon howensis	1	Mature	2018	3	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2000	6	-35.467345	174.740764	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2014	1	-35.47128	174.731295	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2011	2	-37.520791	177.181066	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2016	5	-35.47128	174.731295	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2016	5	-35.47128	174.731295	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2017	4	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2017	4	-35.47128	174.731295	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	1983	8	-35.457412	174.732781	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2018	6	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2018	3	-35.447978	174.731516	WR	Poor Knights Islands

Amphichaetodon howensis	3	Mature	2018	2	-35.56655	174.705996	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2007	8	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	1999	9	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	1999	9	-35.467381	174.736756	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	1999	10	-35.171567	174.331334	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2002	4	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	1992	1	-35.457412	174.732781	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	1997	1	-37.477077	177.133173	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2000	3	-37.287379	176.251751	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2001	3	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2002	9	-35.56655	174.705996	WR	Auckland
Amphichaetodon howensis	2	Unknown	2002	5	-35.56655	174.705996	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2002	5	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2002	5	-35.481915	174.73429	WR	Bay of Plenty
Amphichaetodon howensis	2	Unknown	2002	5	-35.457412	174.732781	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2003	5	-35.56655	174.705996	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2003	5	-35.446198	174.737913	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2003	5	-35.449662	174.740044	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2003	5	-35.547957	174.724348	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2003	5	-35.447351	174.734013	WR	Poor Knights Islands
Amphichaetodon howensis	2	Unknown	2003	5	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Unknown	2004	5	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	Unknown	2004	5	-35.465936	174.73506	WR	Bay of Islands
Amphichaetodon howensis	1	Unknown	2004	5	-35.465936	174.73506	WR	Bay of Islands
Amphichaetodon howensis	1	Unknown	2004	5	-35.485615	174.732551	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2004	5	-35.447351	174.734013	WR	Poor Knights Islands

Amphichaetodon howensis	1	Unknown	2004	5	-35.479993	174.737782	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2010	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2011	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2011	3	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2011	3	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2011	5	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2012	3	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	7	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2012	9	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2012	12	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	juvenile	2019	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	4	Mature	2019	12	-37.287379	176.251751	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2018		-37.287379	176.251751	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2018		-37.287379	176.251751	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2018		-37.287379	176.251751	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2010		-36.524678	175.11492	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	12	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2012	5	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2012	8	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2013	6	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2014	3	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2014	3	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2014	4	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Sub Adult	2016	4	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Sub Adult	2016	5	-35.200302	174.269085	WR	Poor Knights Islands

Amphichaetodon howensis	2	Mature	2012	12	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2013	1	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Unknown	2013	4	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2013	3	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2014	2	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	3	Mature	2014	7	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	1	-35.200302	174.269085	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2015	1	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2017	4	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Sub Adult	2017	4	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2017	8	-35.200302	174.269085	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2010	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2011	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2012	12	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2012	3	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	7	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2012	9	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2013	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2013	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2013	10	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2013	2	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2011	5	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2013	7	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2013	7	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2013	8	-35.447978	174.731516	WR	Poor Knights Islands

Amphichaetodon howensis	1	Mature	2014	1	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2014	10	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2014	10	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2014	11	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2014	12	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2014	8	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2014	5	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2014	5	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2015	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	1	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	1	Mature	2015	11	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	12	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	1	Mature	2015	2	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	3	-35.447978	174.731516	WR	Bay of Islands
Amphichaetodon howensis	2	Mature	2015	3	-35.447978	174.731516	WR	Bay of Plenty
Amphichaetodon howensis	2	Mature	2015	3	-35.447978	174.731516	WR	Bay of Plenty
Amphichaetodon howensis	1	Mature	2015	4	-35.447978	174.731516	WR	Bay of Plenty
Amphichaetodon howensis	1	Mature	2015	4	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	7	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	8	-35.447978	174.731516	WR	Poor Knights Islands
Amphichaetodon howensis	2	Mature	2015	8	-35.447978	174.731516	WR	Bay of Plenty
Amphichaetodon howensis	2	Mature	1980	1	-37.049811	174.674232	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	1990	9	-34.787756	173.377719	WR	Far North
Cheilodactylus ephippium	1	Unknown	2018	3	-35.479909	174.736158	WR	Three kings Islands

Cheilodactylus ephippium	1	Unknown	2014	4	-35.447978	174.731516	WR	Three kings Islands
Cheilodactylus ephippium	1	Unknown	1993	2	-34.151451	172.128555	WR	Three kings Islands
Cheilodactylus ephippium	1	Unknown	1993	2	-34.155263	172.16172	WR	Bay of Plenty
Cheilodactylus ephippium	1	Unknown	1997	3	-34.155263	172.16172	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2000	3	-37.287379	176.251751	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2001	3	-35.447978	174.731516	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2010	2	-35.447978	174.731516	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2011	5	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2014	3	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2014	4	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2015	11	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	2	Mature	2015	3	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2015	5	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2015	9	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2016	1	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2016	1	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	2	Mature	2016	5	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2017	11	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2017	2	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2017	4	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2017	4	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2013	3	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2014	2	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	2	Mature	2015	1	-35.200302	174.269085	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2017	4	-35.200302	174.269085	WR	Poor Knights Islands
Cheilodactylus ephippium	1	Mature	2010	2	-35.447978	174.731516	WR	Bay of Islands

Cheilodactylus ephippium	1	Mature	2013	3	-35.447978	174.731516	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2014	8	-35.447978	174.731516	WR	Bay of Islands
Cheilodactylus ephippium	2	Mature	2015	1	-35.447978	174.731516	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2015	3	-35.447978	174.731516	WR	Bay of Islands
Cheilodactylus ephippium	1	Mature	2016	1	-35.447978	174.731516	WR	Poor Knights Islands
Cheilodactylus ephippium	12	Mature	2019	3	-34.521029	172.964371	WR	Far North
Chrysiptera notialis	1	Mature	2011	4	-35.034245	173.918211	PR	Far North
Chrysiptera notialis	1	Mature	2011	3	-35.194216	174.295452	PR	Bay of Islands
Chrysiptera notialis	1	Juvenile	2019	2	-35.475712	174.73798	EV	Poor Knights Islands
Coryphaena hippurus	1	Unknown	2003	3	-34.410449	173.048083	WR	Far North
Coryphaena hippurus	1	Sub Adult	2003	3	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2003	4	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2002	4	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2002	3	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2002	3	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2003	4	-34.809156	173.399461	WR	Far North
Coryphaena hippurus	1	Unknown	2002	4	-34.999406	173.781402	WR	Far North
Coryphaena hippurus	1	Unknown	2012	4	-34.999406	173.781402	WR	Far North
Coryphaena hippurus	1	Unknown	2016	1	-35.171298	174.330482	WR	Bay of Islands
Coryphaena hippurus	1	Unknown	2014	5	-35.171298	174.330482	WR	Bay of Islands
Coryphaena hippurus	1	Unknown	2019	1	-35.171298	174.330482	WR	Bay of Islands
Coryphaena hippurus	1	Unknown	2016	1	-35.171298	174.330482	WR	Bay of Islands
Coryphaena hippurus	1	Unknown	2014	3	-35.171298	174.330482	WR	Bay of Islands
Coryphaena hippurus	3	Juvenile	2018	2	-35.447351	174.734013	WR	Poor Knights Islands
Coryphaena hippurus	10	Juvenile	2019	2	-35.488367	174.739468	WR	Poor Knights Islands
Coryphaena hippurus	8	Juvenile	2019	3	-35.488367	174.739468	WR	Poor Knights Islands

Coryphaena hippurus	1	Unknown	1991	4	-35.594136	174.618242	WR	Tutukaka
Coryphaena hippurus	1	Unknown	2008	3	-35.594136	174.618242	WR	Tutukaka
Coryphaena hippurus	1	Unknown	2003	4	-35.594136	174.618242	WR	Tutukaka
Coryphaena hippurus	1	Juvenile	2019	2	-35.610347	174.528008	WR	Tutukaka
Coryphaena hippurus	1	Juvenile	2018	2	-36.150953	175.350923	WR	Hauraki Gulf
Coryphaena hippurus	1	Sub Adult	2019	1	-36.414129	176.253122	WR	Coromandel
Coryphaena hippurus	1	Mature	2016	2	-36.450745	175.94136	WR	Coromandel
Coryphaena hippurus	1	Mature	2011	2	-36.797469	175.816481	WR	Coromandel
Coryphaena hippurus	1	Sub Adult	2017	2	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Mature	2016	2	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Mature	2016	2	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	2	Sub Adult	2019	1	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	3	Sub Adult	2019	2	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Mature	2015	3	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Unknown	2015	3	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Unknown	2015	2	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Sub Adult	2016	1	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Mature	2016	1	-36.797481	175.808243	WR	Coromandel
Coryphaena hippurus	1	Unknown	2004	2	-36.821547	175.756189	WR	Coromandel
Coryphaena hippurus	1	Mature	2016	2	-37.004643	176.841352	WR	Coromandel
Coryphaena hippurus	1	Mature	2019	3	-37.123067	174.430405	WR	Manukau
Coryphaena hippurus	1	Mature	2018	1	-37.123067	174.430405	WR	Manukau
Coryphaena hippurus	1	Unknown	2004	2	-37.377158	178.298857	WR	East Cape
Coryphaena hippurus	1	Unknown	2015	2	-37.520791	177.181066	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2014	3	-37.520791	177.181066	WR	Far North
Coryphaena hippurus	1	Unknown	2011	2	-37.520791	177.181066	WR	Bay of Plenty

Coryphaena hippurus	1	Unknown	2011	3	-37.520791	177.181066	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2011	3	-37.520791	177.181066	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2011	3	-37.520791	177.181066	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2011	3	-37.520791	177.181066	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2004	2	-37.531797	176.703124	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2018	2	-37.536177	177.981482	WR	East Cape
Coryphaena hippurus	1	Unknown	2007	4	-37.592872	177.936482	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2006	2	-37.651999	176.165167	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2008	3	-37.790492	174.909823	WR	Raglan
Coryphaena hippurus	1	Unknown	2010	2	-37.790492	174.909823	WR	Raglan
Coryphaena hippurus	1	Unknown	1985	3	-37.933883	177.014288	WR	Bay of Plenty
Coryphaena hippurus	1	Unknown	2001	2	-37.933883	177.014288	WR	Bay of Plenty
Coryphaena hippurus	1	Sub Adult	2018	2	-38.983252	174.231894	WR	Taranaki
Coryphaena hippurus	1	Sub Adult	2018	2	-38.983252	174.231894	WR	Taranaki
Coryphaena hippurus	40	Sub Adult	2016	4	-39.276968	173.745829	PR	Taranaki
Coryphaena hippurus	50	Mature	2017	4	-39.539477	177.794834	RE	Hawke’s bay
Epinephelus daemelii	1	Sub Adult	2007	3	-35.594136	174.618242	WR	Bay of Islands
Epinephelus daemelii	1	Sub Adult	2014	1	-35.47128	174.731295	WR	Bay of Plenty
Epinephelus daemelii	1	Unknown	2011	5	-40.85676	174.912292	WR	Far North
Epinephelus daemelii	1	Mature	2010	4	-34.155263	172.16172	WR	Far North
Epinephelus daemelii	1	Sub Adult	1986	12	-34.410449	173.048083	WR	Far North
Epinephelus daemelii	1	Mature	1994	4	-34.797511	173.411827	WR	Far North
Epinephelus daemelii	3	Mature	2001	4	-35.197072	174.299185	WR	Far North
Epinephelus daemelii	5	Unknown	1978	6	-35.437907	174.40639	WR	Kapiti Island
Epinephelus daemelii	2	Unknown	1993	2	-34.151451	172.128555	WR	Poor Knights Islands
Epinephelus daemelii	2	Unknown	1993	2	-34.155263	172.16172	WR	Poor Knights Islands

Epinephelus daemelii	1	Unknown	1993	2	-34.155263	172.16172	WR	Poor Knights Islands
Epinephelus daemelii	2	Unknown	1993	2	-34.155263	172.16172	WR	Poor Knights Islands
Epinephelus daemelii	1	Unknown	1993	2	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	1	Unknown	1993	2	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	2	Unknown	1993	2	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	1	Unknown	1993	2	-34.151451	172.128555	WR	Three kings Islands
Epinephelus daemelii	2	Unknown	1997	3	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	1	Unknown	1997	3	-34.151451	172.128555	WR	Three kings Islands
Epinephelus daemelii	2	Unknown	1997	3	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	2	Unknown	1997	3	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	1	Unknown	1997	3	-34.155263	172.16172	WR	Three kings Islands
Epinephelus daemelii	1	Mature	2011	3	-35.447978	174.731516	WR	Three kings Islands
Epinephelus daemelii	1	Juvenile	2019	2	-34.915267	173.558883	WR	Three kings Islands
Epinephelus daemelii	2	Mature	2018	11	-37.447085	176.331967	WR	Three kings Islands
Epinephelus daemelii	1	Sub Adult	2011	3	-35.447978	174.731516	WR	Three kings Islands
Epinephelus daemelii	1	Sub Adult	2015	5	-35.447978	174.731516	WR	Three kings Islands
Epinephelus daemelii	2	Mature	2019	3	-34.516437	172.923057	WR	Tutukaka
Epinephelus daemelii	1	Juvenile	2019	2	-35.83375	174.47061	WR	Whangarei
Epinephelus daemelii	1	Sub Adult	2019	1	-34.945797	173.653737	WR	Far North
Epinephelus daemelii	1	Mature	2019	1	-34.945797	173.653737	WR	Far North
Parablennius intermedius	1	Mature	2019	3	-34.534594	172.944034	RE	Far North
Parablennius intermedius	2	Mature	2019	3	-34.534594	172.944034	RE	Far North
Pomatomus saltatrix	1	Unknown	2012	12	-34.997889	173.964701	N/A	Far North

3.8.2 Appendix 2: Supplementary methods and resources for ten target species from the case-study

Here we describe in detail our classification of the citizen science observations and the contribution by the expert panel for our target species to demonstrate our stepwise approach with reference to the main text and other supporting information. These supplementary resources are intended to provide a guide for applying our method to alternate target species.

Expert panel

An expert panel was formed to meet for a one-day workshop to inform decisions for Step1, 2 and 5. Panel members consisted of New Zealand-based science professionals including: marine scientists (four individuals, with a combined total of 350+ peer-reviewed publications on species biology, marine biosecurity and population dynamics in New Zealand and the South Pacific), experienced taxonomists (two individuals with >15 years' experience each and >1200 combined citations) and resource managers (one individual with over 20 years' experience in fisheries regulation) representing eleven research institutions including universities, museums and government agencies. The panel also included experienced citizen scientists consisting of: spearfishers (three active individuals with recognised observational expertise and over 50 years' combined experience), fishers (two active individuals that represent both commercial and recreational sectors and engage through published media), and SCUBA divers (three experienced divers and underwater photographers with recognised observational expertise and over 50 years combined experience).

During the one-day workshop the panel was broken into three sub-groups and assigned a list of target species to work through Steps 1, 2 and 5. The sub-groups were organised based on expertise (i.e., Fishing, SCUBA, Spearfishing) and were assigned species according to the mode by which the species was typically observed (i.e., fishing: *species sighted was predominantly caught by commercial or recreational fishers*, or underwater visual: *species predominantly sighted in-situ by SCUBA divers, snorkellers, freedivers and spearfishers*). Several species were assigned across multiple sub-groups to determine if there were sub-group biases and to increase overall confidence. All Step outcomes were reviewed by the entire panel and final decisions were made by group consensus.

Case-study: Suspected out-of-range marine fishes in Aotearoa New Zealand

From the species pool assessed by the expert panel, we selected ten target species to provide a casestudy that demonstrated the utility of the classification tools. The source database for all fishes presented to the expert panel was constructed using two peer-reviewed publications that comprise a comprehensive list of New Zealand's marine fishes, i.e., *The Fishes of New Zealand* (Roberts et al., 2015) and updates (Roberts et al., 2017, 2019) as well as *Coastal Fishes of New Zealand* (Francis, 2012). We searched these publications for marine ray-finned fishes (Class Actinopterygii) using the following keywords: invasive', 'alien' 'vagrant', 'occasional', 'tropical', 'subtropical', 'rare' or 'new-to-New Zealand'. Any species identified by those keywords in their range, distribution, abundance or affinity was included in the database.

The ten target species were selected to span a broad range of habitats, sizes, dietary preferences, levels of mobility, range sizes and latitudinal ranges (Table S2.1) to test the versatility of the classification tools. Observations of these target species were collated from a broad range of data sources including peer-reviewed literature (published manuscripts, checklists and books), unpublished occurrence records held by scientists, underwater observations, photographic records and other unpublished anecdotal information obtained from citizen scientists. Data sources varied in type, content, completeness and validity. For example, data from social media and online forums were complex, unstructured and of variable quality, therefore an observation validation method was developed to maintain a level of consistency between disparate data sources (Figure S2.1). The validation method required a higher level of completeness for observations made by citizen scientists versus those made by expert scientists. Only observations that met a certain level of completeness and quality were included in the case-study dataset (Table S2.1).

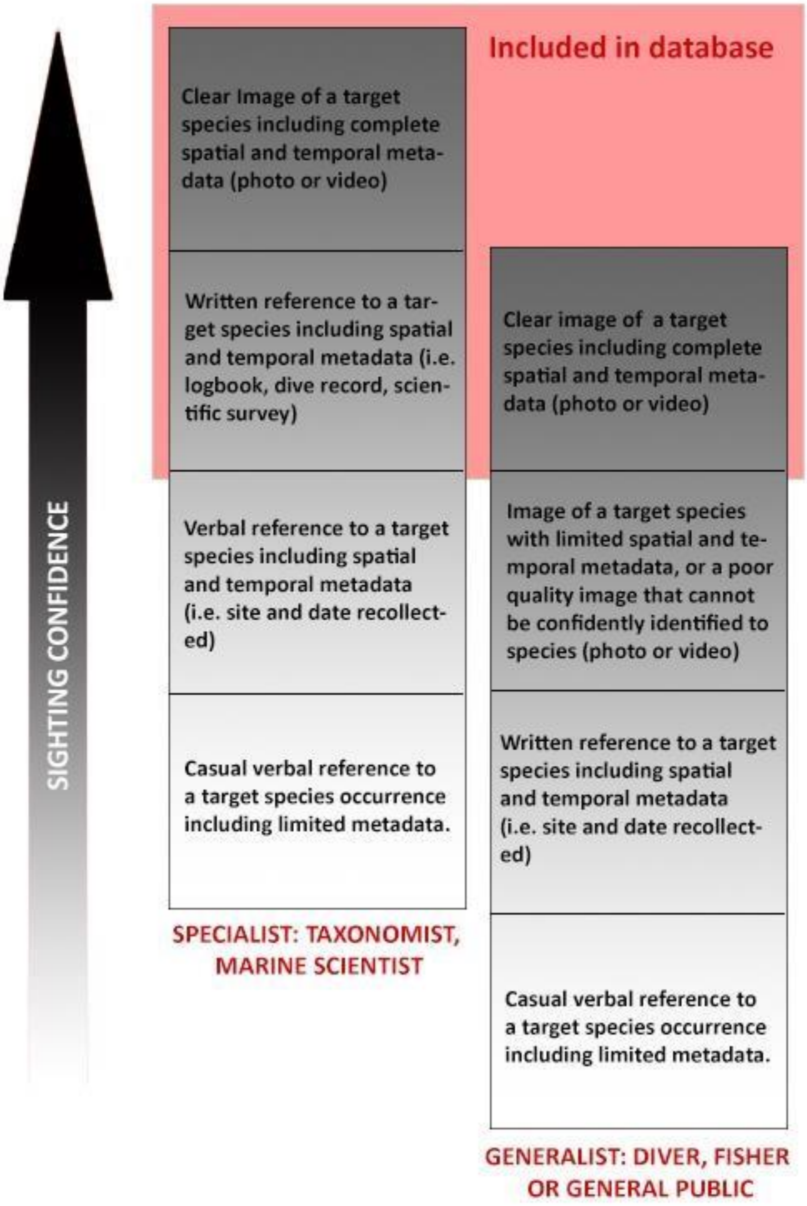


Figure S2.1: Observation validation criteria for inclusion. A higher level of observation confidence is required for observations by citizen scientists versus expert scientist observations.

Step 1: Determining the species spatio-temporal range and expert confidence

The first stage of our method used a decision tree to help describe the spatio-temporal range (i.e., extent of the species' typical geographic distribution for a given season/s) for each target species with specific reference to New Zealand. We then determined the collective confidence of an expert panel in the defined species' range/s. For each target species, the expert panel first answered Question 1.1 (Figure S2.2); whether there was any taxonomic or identification confusion that could affect our ability to accurately define the species' range. For instance, in cases where the target species could be confused with a similar species that had an overlapping or adjacent geographic distribution, or there had been taxonomic or nomenclature changes that may undermine knowledge of the species' range, the panel answered 'YES' and moved to Question 1.2b. If the panel was confident there was no taxonomic or nomenclature confusion relating to the target species that would influence the definition of the range, they answered 'NO' to Question 1.1 and moved to 1.2a.

Table S2.1: Target species traits, geographic affinities, and habitats.

Species	**Distribution**[1]	**Max size (mm)**[1]	**Swim Type**[1]	**Aggregation type**[2]	**Habitat**[1]	**Diet**[1]	**Trophic level**[1]	**Breeding mode**[1]	**Spawning mode**[1]	**Temperature range**[3]	**Latitudinal range**[1,3]	**Depth range**[1]
Abudefduf vaigiensis	Tropical	200	Labriform	Schooling	Reef	Omnivore	2.6	Paired	Nesting	15-30°C	36°N-39°S	1-15m
Acanthocybium solandri	Subtropical	2500	Thunniform	Solitary	Pelagicoceanic	Carnivore	4.5	Group	Broadcast	10-30°C	59°N-48°S	0-20m
Acanthurus dussumieri	Tropical	540	Labriform	Schooling	Reef	Herbivore	2	Paired	Broadcast	24-28°C[1]	40°N-37°S	4-131m
Amphichaetodon howensis	Temperate	180	Balistiform	Pairs	Reef	Omnivore	*Unknown*	Paired	Nonguarder	15-25°C	21°S-38°S	10-150m
Cheilodactylus ephippium	Subtropical	350	*Unknown*	Pairs	Reef	Carnivore	3.5	*Unknown*	*Unknown*	15-25°C	30°S-37°S	0-250m
Chrysiptera notialis	Subtropical	90	Labriform	Pairs	Reef	Omnivore	2.7	Paired	Nesting	20-25°C	17°S-28°S	7-45m
Coryphaena hippurus	Tropical/Subtropical	2100	Thunniform	Schooling	Pelagicneritic	Carnivore	4.5	Group	Broadcast	21-30°C	47°N-40°S	0-85m
Epinephelus daemelii	Subtropical	2000	*Unknown*	Solitary or Pairs	Reef	Carnivore	4.5	Paired	Broadcast	*Unknown*	25°S-43°S	0-50m
Parablennius intermedia	Subtropical	120	*Unknown*	Solitary	Reef and Brackish	Carnivore	3.4	Paired	Nesting	15-30°C	7°S-35°S	1-10m
Pomatomus saltatrix	Subtropical	1300	Carangiform	Schooling	Pelagicoceanic	Carnivore	4.5	Group	Broadcast	10-30°C	45°N-44°S	0-200m

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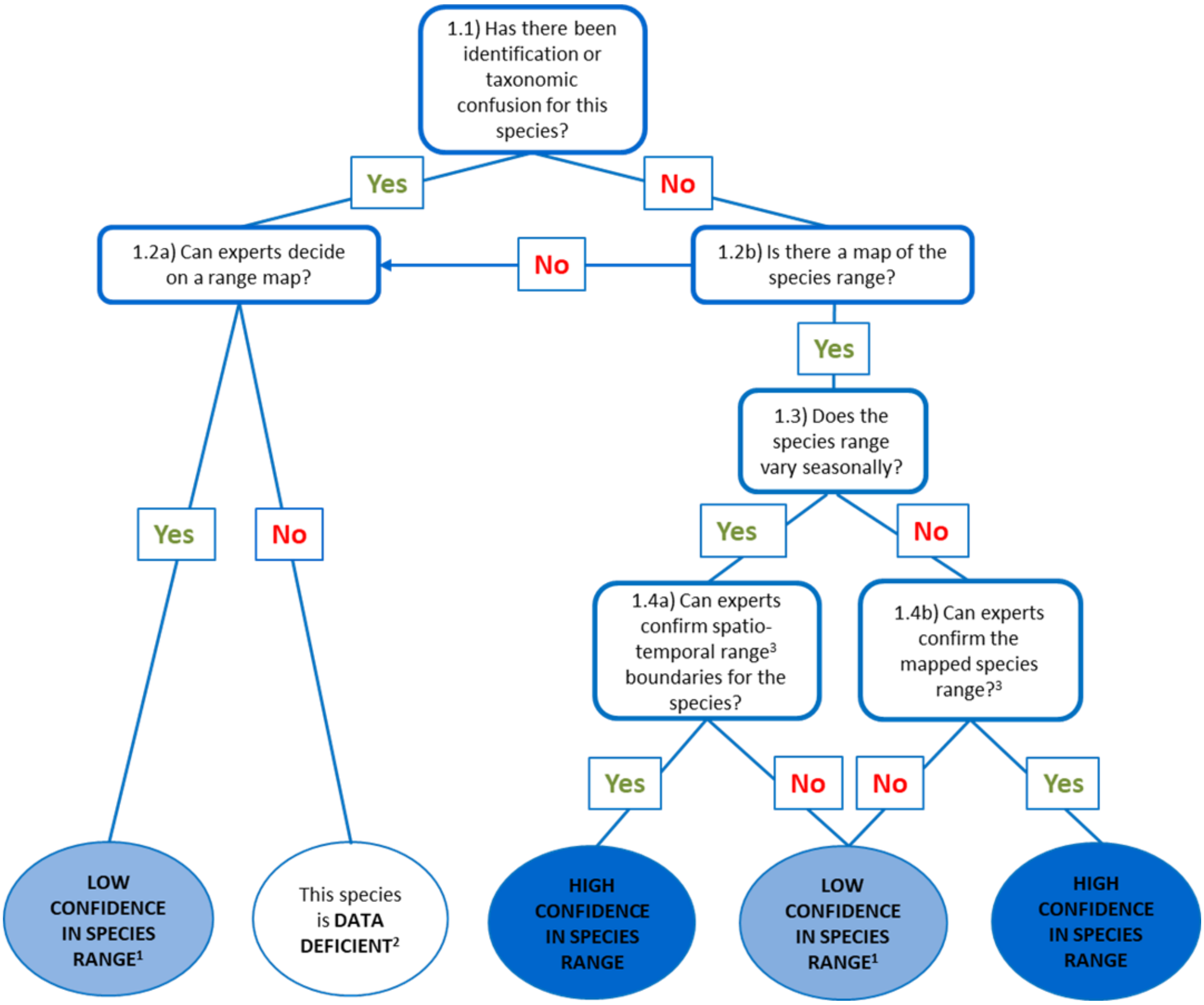
[1] Fishbase. Froese, R. and D. Pauly. Editors. (2019). FishBase. World Wide Web electronic publication. www.fishbase.org, version (12/2019).

[2] Roberts, C., Stewart, A. L., & Struthers, C. D. (2015). The Fishes of New Zealand (C. D. Roberts, A. L. Stewart, & C. D. Struthers Eds.): Te Papa Press.

[3] OBIS (2020) Ocean Biogeographic Information System. Intergovernmental Oceanographic Commission of UNESCO. www.iobis.org.



[1] If the confidence in a species spatio-temporal range was 'LOW', obtaining more occurrence records could assist in increasing the confidence.

[2] If a species was classified as data deficient, further research should focus on morphology and genetic identification tools and/or species distribution surveys. [3] In Step 1.4 the expert panel is asked to confirm the current species mapped range based on an independent or corroborating secondary information source, such as quantitative survey data or unpublished knowledge.

Figure S2.2. Decision tree for determining the spatio-temporal range of a species (*i.e.*, *extent of the species typical geographic distribution for a given season/s*) and the confidence of the expert panel in the mapped species' range.

To assist the expert panel in addressing Question 1.2a, the authors constructed putative range maps using data sourced from the Ocean Biogeographic Information System (OBIS; www.iobis.org). OBIS is an online open-source data platform that collates data from various sources including museum collections, research institutes and universities, to provide abundance and distributional information for marine species. Using OBIS data to create range maps can underestimate or overestimate species' distributions, as some of the observations may be extralimital vagrants (i.e., *a juvenile individual observed outside of its known range)* or recording errors. Therefore, the expert panel verified and refined the OBIS range maps according to published resources (including Roberts et al., 2017; Francis, 2012; and peer-reviewed literature) and their own observations and knowledge in assessing Question 1.2a, the before moving to Question 1.3. If OBIS observational data were not available for a given species, we attempted to source at least two range maps or descriptions from published sources; verification and refinement of these maps proceeded as described previously before moving to Question 1.3. If no mapped range was able to be sourced the assessment moved to Question 1.2a.

There were two reasons why the assessment would move to Question 1.2b; first, if there was a level of taxonomic confusion in Question 1.1, or if there is no mapped or quantitative assessment of the species' range available in Question 1.2a. Question 1.2b was used to determine if the panel could describe a qualitative range for the species by consensus, this description could include the following: identification of the range limits, the areas or biogeographic regions where the species was present, and/or the latitudinal extent of the species' range. If a qualitative range could be described, this resulted in a 'LOW' confidence in the species' range; if a qualitative assessment of the species' range was unavailable the species was not assessed further and was identified as Data Deficient.

If the panel determined that the target species' range varied seasonally (Question 1.3; i.e., expected for several pelagic species) they were prompted to describe the spatio-temporal range of this species in Question 1.4a. If the target species showed no seasonal variability ('NO' in Question 1.3) the assessment proceeded to Question 1.4b, where the accuracy of the species' spatial range boundaries was determined.

If the expert panel confirmed the accuracy of either the species' spatio-temporal or spatial range (Question 1.4a and 1.4b, respectively) the confidence in the species' range accuracy was assigned as 'HIGH'. If there was ambiguity around the species spatio-temporal or spatial range this resulted in 'LOW' confidence in the accuracy of the species' range. To summarise, the outcomes of Step 1 included a species' spatiotemporal range map/s and an associated measure of confidence in that range.

Step 2: Scoring species detectability

Detectability of a species by citizen scientists or other observers is a function of its natural abundance, habitat preference, morphology and behaviour (Robinson et al., 2015). The first step in determining species detectability for our assessment was identifying the predominant method of detection for the target species: fishing (*species predominantly caught by commercial or recreational fishers*) or underwater visual (*species predominantly sighted in-situ by SCUBA divers, snorkellers, freedivers and spearfishers*). Next, the abundance of the target species was determined based on how frequently (rarely = 1, sometimes = 2, or commonly = 3) the primary habitat a species occupies is fished or observed, and the abundance of the species within those habitats for their known range (rare = 1, patchy = 2, or common = 3), these were combined to give an overall abundance score. If the combined score was five or more, a species was considered 'abundant' if the score was less than five, a species was considered 'not abundant'. (Table S2.2). If target species were usually detected by underwater visual observations, we additionally scored the conspicuousness of mature and juvenile life-stages of that species. Our method of scoring conspicuousness was consistent with that of Robinson et al. (2015); species were scored as 'conspicuous' or 'inconspicuous' based on the maximum adult body size, the species' ability to camouflage and if it concealed itself or routinely hides within structure or substrate (maximum size 30cm or larger = 1, does not camouflage = 1, and does not hide = 1). These three factors would influence the ability of observers to detect individuals *in-situ*, and have been used in previous studies of species observations (Bayley & Peterson, 2001; Bozec et al., 2011). A species was deemed conspicuous if the total score was greater than, or equal to two; if the score was less than two it was deemed inconspicuous. For fishing species, detectability was 'HIGH' if a species was abundant; for underwater visual species, detectability was 'HIGH' if the species was either conspicuous or abundant, as these were considered equally important for *in-situ* detection.

Table S2.2. Detectability scoring method (Step 2). (a) Abundance scoring for species observed predominantly by *fishing* and *underwater visual* observation, and (b) Conspicuousness scoring for species detected using *underwater visual* observation. A species (or life-stage) was deemed 'conspicuous' if the total score was greater than or equal to two; if the score was less than two it was deemed 'inconspicuous'. The detection method, abundance and conspicuousness scores were combined to inform an overall level of 'detectability' (c) for both mature and juvenile life stages of target species.

(a) Species abundance	**Score**
How commonly is the species habitat observed[1] or fished?[2]	
Rarely	1
Sometimes	2
Commonly	3
How common is the species in its habitat?[3]	
Rare	1
Patchy	2
Common	3

(b) Species conspicuousness	**Score**
Camouflage[4]	
Species exhibits some form of camouflage	0
Species does not exhibit camouflage	1
Body Size[5]	
Maximum adult body size smaller than 30cm	0
Maximum adult body size 30cm or larger	1
Hiding[6]	
Species hides behind structures or within substrate	0
Species does not hide	1

(c) Detectability scoring			
Detection method	*Abundance (a)*	*Conspicuousness (b)*	*Detectability*
Underwater visual	Abundant	Conspicuous	HIGH
Underwater visual	Abundant	Inconspicuous	HIGH
Underwater visual	Not abundant	Conspicuous	HIGH
Underwater visual	Not abundant	Inconspicuous	LOW
Fishing	Abundant	NA	HIGH
Fishing	Not abundant	NA	LOW

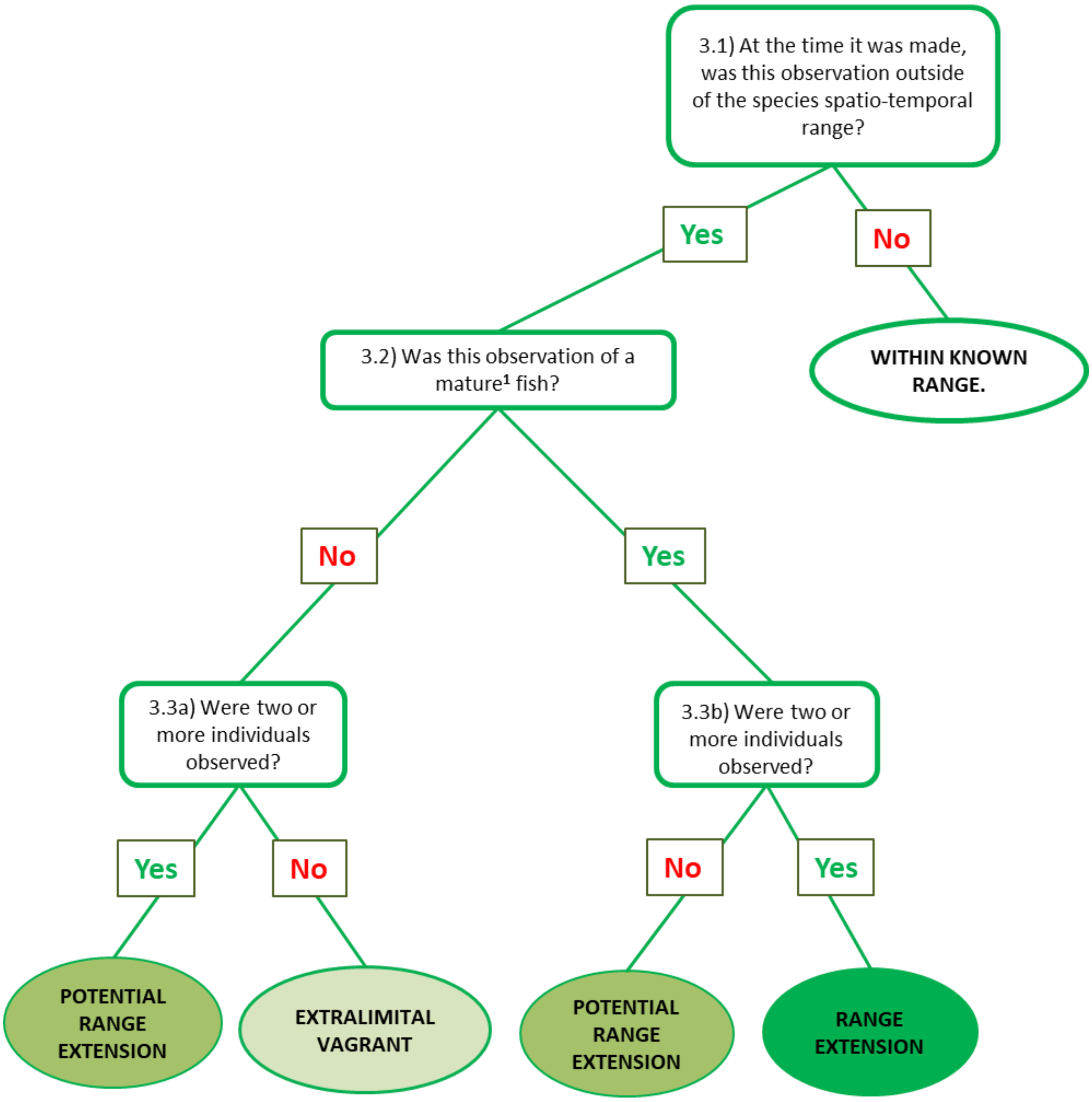
[1] in areas that observers in New Zealand may dive, spearfish, or swim.
[2] including recreational and commercial fishing in New Zealand waters. [3] within the species' known range.
[4] camouflage includes background matching, self-shadow concealment, obliterative shading (countershading), and masquerade (leaf/other fish). [5] body size was obtained from peer-reviewed literature, online databases, or expert knowledge.
[6] if a species is known to hide, i.e., it generally conceals itself behind an object or inside cracks and crevices. Behavioural data were obtained from peer-reviewed literature or expert knowledge.

Step 3: Classifying citizen science observations of species

In Step 3 of our method, a decision tree was used to classify each citizen science observation of a target species as one of the following: within known range (WR), extralimital vagrancy (EV), possible range extension (PR) or range extension (RE). The first consideration of this decision tree was to determine if the species had been sighted outside of the expert-defined spatio-temporal range identified in Step 1. If the answer to Question 3.1 was 'NO', this observation was not assessed further. If the observation was outside of the spatio-temporal range, i.e., an answer of 'YES' for Question 3.1 the assessment progressed to Question 3.2 (Figure S2.3).

Question 3.2 asked whether the observations were of mature individuals; features of maturity may vary between species but could include being of or near maximum size or reproductive size, mature colouration, or spawning or nesting behaviour. If a species has reached maturity or a large size by the time it was sighted, it could have survived through winter or multiple years prior to being sighted and is also more likely to be able to successfully reproduce and establish a new population. Assessing maturity required an estimate of the size of the individuals sighted or behavioural observations; if these data were not available for an observation a conservative approach was taken and the answer to Question 3.2 was 'NO'.

The number of individuals per observation is assessed in Question 3.3a and 3.3b; if more than two individuals were sighted at the same location at the same time, there is a higher probability of the species being able to reproduce, establish a self-recruiting population and persist. If more than two juvenile fishes were sighted outside of their spatio-temporal range it was classed as a potential range extension and if only one juvenile individual was sighted outside of the spatio-temporal range it was classed as an extralimital vagrant. If a single mature fish was sighted outside of its spatio-temporal range it was classed as a potential range extension, as this individual could have overwintered and may have the potential to reproduce if it encounters another individual. If multiple mature fishes were sighted outside of the spatiotemporal range at the same location at the same time, the observation was designated as a range extension. Step 3 resulted in a classification for each observation of a target species; for some species this may only be a single data point and for others, there may be a range of different classifications for a single species, due to the spread of the observations.



[1] The features used to determine maturity in Step 3.2 may vary among species but are likely to include reaching maximum size and/or reproductive size, mature colouration or spawning/nesting behaviour.

Figure S2.3. Decision tree for classifying species observations as within known range, extralimital vagrancy, potential range extension, or range extension using the spatio-temporal range determined by the expert panel in Step 1.

Step 4: Determining confidence in classification of observations

In some applications, like invasive species management or when allocating resources to species management, a score of 'overall confidence' in the classification of observations may be required. In Step 4 we combined the confidence in species' range from Step 1 and species detectability from Step 2 to determine 'overall confidence'. Increased confidence in a species' spatio-temporal range gives greater confidence in the identification of out-of-range occurrences. However, because species detectability influences the likelihood that our knowledge accurately represents the species current distribution, species detectability also contributes to our overall confidence (Table S2.3). Species' range confidence (Step 1) and species detectability scores (Step 2) were fed into a confidence matrix to provide an overall confidence in the classification of each observation for that species.

Table S2.3. Confidence scoring method for the classification of observations (Step 4). Overall confidence scores are a function of the spatio-temporal range confidence (Step 1) and detectability

(Step 2). Range confidence can vary seasonally (i.e., a summer range and a winter range) and/or across life-stages (i.e., juvenile and mature).

		Step 1: Spatio-temporal range confidence	
		LOW	*HIGH*
Step 2: Species detectability	*LOW*	**NO** Confidence	**MODERATE** Confidence
	HIGH	**LOW** Confidence	**HIGH** Confidence

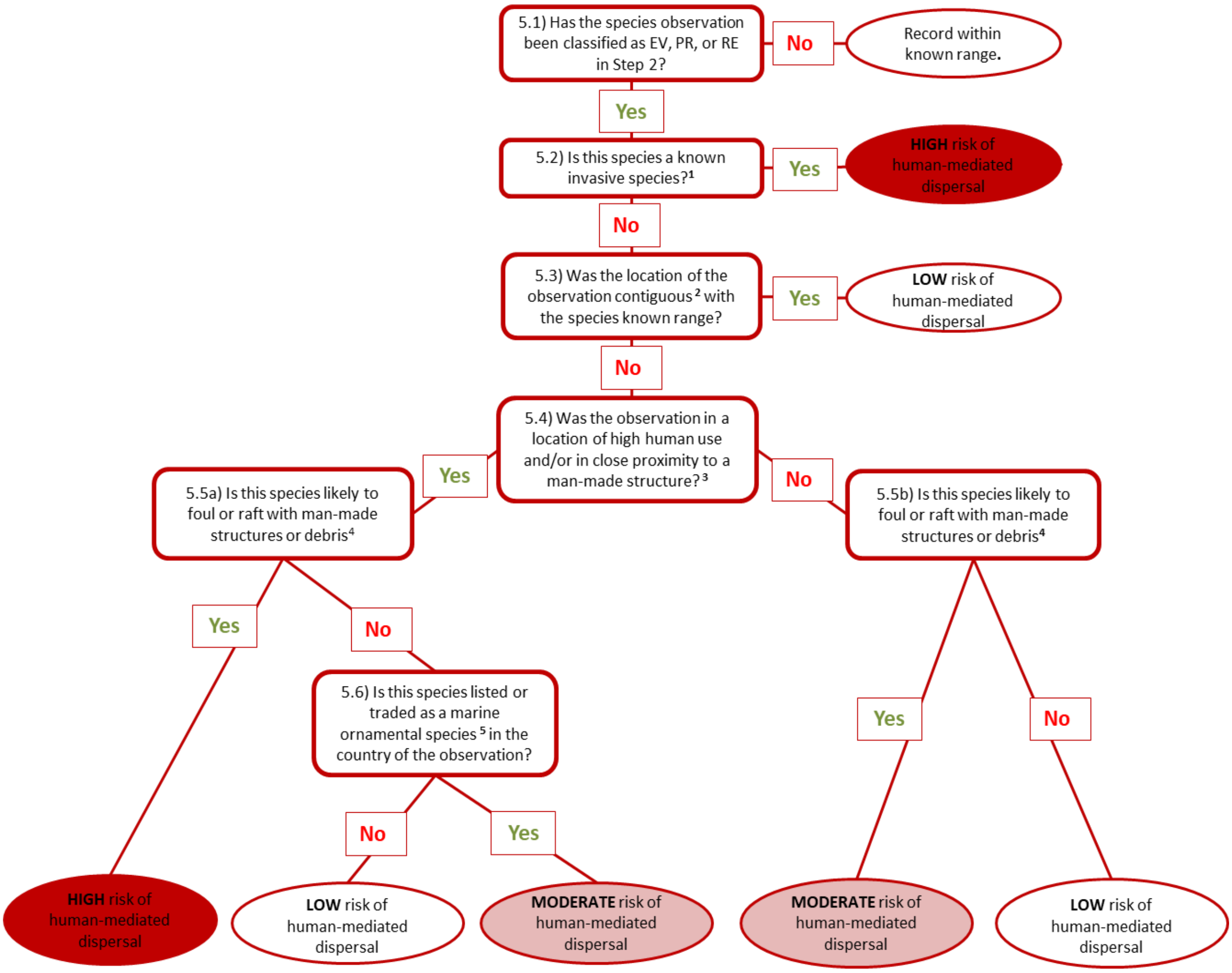
Step 5: Determining confidence that out-of-range occurrences are due to human-mediated dispersal

For out-of-range occurrences from Step 3, we assigned a level of risk (‘LOW’, ‘MODERATE’ or ‘HIGH’) that they may be as a result of human-mediated dispersal using a decision tree (Figure S2.4). In Question 5.1 we asked if species observations were identified as an extralimital vagrant, potential range extension or range extension in Step 3. If the answer was ‘YES’ the assessment moved to Question 5.2; here we ask if the species was recognised as a known invasive species. If the species was listed nationally or locally as a pest, invasive or unwanted species or has been identified as an invasive species in peer-reviewed literature or published reports, the risk was ‘HIGH’ that it may have been spread by human-mediated pathways. If the species was not a known invasive species the assessment moved to Question 5.3, where we assessed the contiguity of the species’ known range.

Source populations for marine invasive species are often distant from the recipient communities and are transported great distances, across natural dispersal barriers, by human-mediated vectors. If the species’ range was contiguous there was a ‘LOW’ risk that the out-of-range occurrence was due to human mediated dispersal, as it is likely to naturally spread to neighbouring areas. If the species’ range was disjunct the assessment moved to Question 5.4. This Question assessed the proximity of the observation to areas of high human use or man-made structures. Human mediated introductions are more likely in areas of high human use or population density; these are often hubs of vessel movements and species held by people in aquaria. If the observation was near a population hub, the answer to Question 5.4 was ‘YES’ and the assessment progressed to Question 5.5a. If the answer was ‘NO’ i.e., the observation was away from any area of high human use or man-made structure, the assessment moved to Question 5.5b. In both Question 5.5a and 5.5b we ask if the species has a propensity to foul vessels or raft with man-made debris; if the answer to Question 5.5b was ‘YES’ the risk was ‘MODERATE’ that it may have been spread by human-mediated pathways. If the observation was distant from any areas of high human use and is unlikely to raft or foul vessels (i.e.,‘NO’ to Question 5.5b), the risk was ‘LOW’ that the observation was facilitated by human mediated vectors.

If the assessment had progressed to Question 5.5a and the answer was ‘YES’, the combination of the species being in an area of high human use or population density and being likely to raft or foul man-made structures increased the risk of the species being introduced by human mediated vectors to ‘HIGH’. If the answer to Question 5.5a was ‘NO’, we assessed if the species is likely to be traded as a marine ornamental species, and if it is on a local ornamental species import list. The combined risk from the fact the species was in an area of high human use and can be kept as an aquarium species suggests it could have been released by a member of the public either intentionally or accidentally i.e., if the answer is 'YES' to

Question 5.6, the risk was 'MODERATE' that it may have been spread by human-mediated pathways. If the answer to Question 5.6 was 'NO', there was a 'LOW' risk that the out-of-range occurrence was due to human-mediated dispersal.



[1] Classified as an unwanted organism or pest by a regional authority (e.g., Biosecurity New Zealand), or listed on an international unwanted species register such as the Global Invasive Species Database or the National Exotic Marine and Estuarine Species Information System (NEMESIS) database. [2] A contiguous range is continuous spatially and not patchy or geographically disjunct. [3] Such as wharves, marinas, marine farms, mooring areas and harbours in, and adjacent to, urban areas. [4] Such as floating plastic, fishing debris, macroalgal rafts, terrestrial plant material, vessels and marine structures. [5] Such as the New Zealand Ministry for Primary Industries "Schedule 4 Approved species of ornamental fish and marine invertebrates" (document ORMAMARI.ALL, 13 April 2017).

Figure S2.4. Decision tree for determining confidence in out-of-range observations classified in Step 3 (extralimital vagrant, possible range extension, or range extension) as human-mediated.

Supplementary resources

We used several reference materials to assist in the stepwise assessments and to inform the expert panel decisions. The following reference materials were made available to the expert panel during the workshop to assist in the definition of species spatio-temporal ranges (Step 1), detectability (Step 2), and our confidence that an out-of-range occurrence may be due to human-mediated dispersal (Step 5).

Bay of Plenty regional council, Pest Management Strategy: https://www.boprc.govt.nz/environment/pests/marinepests/

Hayes, K. R., McEnnulty, F. R., & Sliwa, C. (2002). Identifying Potential Marine Pests - An Inductive Approach. Centre for Research on Introduced Marine Pests.

ICUN Invasive Species Specialist Group, Global Invasive Species Database: http://www.iucngisd.org/gisd/
IUCN Invasive Species Specialist Group, World Register of Introduced Species (WRiMS): http://www.marinespecies.org/introduced/

MPI, register for unwanted and notifiable organisms: https://www.mpi.govt.nz/protection-and-response/finding-andreporting-pests-and-diseases/registers-and-lists/

MPI, Marine pest ID guide: https://www.mpi.govt.nz/dmsdocument/10478/direct

Northland regional council, Regional Pest Management Strategy: https://www.nrc.govt.nz/resource-librarysummary/plans-and-policies/pest-management/

Smithsonian Environmental Research Centre, National Exotic Marine and Estuarine Species Information System: https://invasions.si.edu/nemesis/

Top of the North Marine Biosecurity Group: https://marinepests.nz/

Ayling, T., & Cox, G. J. (1982). Collins guide to the sea fishes of New Zealand, . Auckland Collins.

FishBase. (2019). FishBase. version (12/2019). Retrieved from www.fishbase.org

Francis, M. P. (1988). Confirmation of Parma polylepis, a pomacentrid teleost, in New Zealand waters (note). New Zealand Journal of Marine and Freshwater Research, 22(1), 143-145.

Francis, M. P. (1993). Further Aadditions to the fish faunas of Lord Howe and Norfolk Islands, Southwest Pacific Ocean. Pacific Science, 47(2), 118-135.

Francis, M. P. (1996). Geographic distribution of marine reef fishes in the New Zealand region. New Zealand Journal of Marine and Freshwater Research, 30(1), 35-55.

Francis, M. P. (2012). Coastal Fishes of New Zealand (Vol. 4). Nelson: Craig Potton Publishing.

Francis, M. P., & Duffy, C. A. (2015). New records, checklist and biogeography of Kermadec Islands' coastal fishes. In T. Trnski & H. A. Schlumpf (Eds.), Bulletin of the Auckland Museum (Vol. 20, pp. 481-495). Auckland.

Francis, M. P., & Evans, J. (1992). Immigration of subtropical and tropical animals into north-eastern New Zealand. Paper presented at the Second International Temperate Reef Symposium, University of Auckland.

Francis, M. P., Worthington, C. J., Saul, P., & Clements, K. D. (1999). New and rare tropical and subtropical fishes from northern New Zealan. New Zealand Journal of Marine and Freshwater Research, 33(4), 571-586.

Hayes, K. R., McEnnulty, F. R., & Sliwa, C. (2002). Identifying Potential Marine Pests - An Inductive Approach. Centre for Research on Introduced Marine Pests.

McMillan, P. J., Francis, M. P., James, G. D., Paul, L. J., Marriott, P. J., Mackay, E., . . . Wei, F. (2011). New Zealand Fishes Volume 1: A field guide to common species caught by bottom and midwater fishing. Wellington: NIWA.

OBIS. (2020). Ocean Biogeographic Information System.. Retrieved from www.iobis.org

Pharo, H. (2017). Import Health Standard: Ornamental Fish and Marine Invertebrates. Ministry for Primary Industries. Wellington.

Roberts, C., Stewart, A., Struthers, C., Barker, J., & Kortet, S. (2017). Checklist of the fishes of New Zealand: Online version 1.0. Museum of New Zealand Te Papa Tongarewa, 1.0, 1-176.

Roberts, C., Stewart, A. L., & Struthers, C. D. (2015). The Fishes of New Zealand (C. D. Roberts, A. L. Stewart, & C. D. Struthers Eds.): Te Papa Press.

3.8.3 Appendix 3: Supplementary results from Step 1 and 2 of the case-study for all target species

Table S3.1 Results for Step 1: Determining the species spatio-temporal range and expert confidence as determined by the expert panel by consensus for all target species assessed during the one-day workshop.

Species name	**Detection method** (Underwater visual V; Fishing F)	**Range confidence** (High H; Low L; Data Deficient DD)	**Why was the species Data Deficient?** (Taxonomic confusion 'YES' Question 1.1 Figure 2, No range map: 'NO' Question 1.2a Figure 2)
Acanthistius cinctus	V	L	N/A
Acanthocybium solandri	F	DD	No Range map
Acanthopagrus australis	F	DD	No Range map
Acanthopagrus pacificus	F	L	N/A
Acanthurus dussumieri	V	L	N/A
Aluterus scriptus	V	DD	No Range map
Amphichaetodon howensis	V	H	N/A
Anampses elegans	V	H	N/A
Antennarius commerson	V	DD	Taxonomic confusion
Antennarius striatus	V	L	N/A
Aplodactylus etheridgii	V	H	N/A
Arothron firmamentum	V	L	N/A
Arripis xylabion	F	L	N/A
Atypichthys latus	V	H	N/A
Aulacocephalus temmincki	V	H	N/A
Aulostomus chinensis	V	L	N/A
Bodianus bimaculatus	V	DD	No Range map
Bodianus flavifrons	F	L	N/A

Bodianus flavipinnis	F	L	N/A
Cantherhines dumerili	V	DD	No Range map
Canthigaster callisterna	V	H	N/A
Cheilodactylus ephippium	V	H	N/A
Cheilodactylus fuscus	V	L	N/A
Cheilodactylus nigripes	V	H	N/A
Chilomycterus reticulatus	V	DD	Taxonomic confusion
Chromis abyssicola	V	DD	Taxonomic confusion
Chromis flavomaculata	V	L	N/A
Chromis fumea	V	H	N/A
Chrysiptera notialis	V	DD	No Range map
Coris picta	V	H	N/A
Coryphaena hippurus	F	H	N/A
Cyprinocirrhites polyactis	V	DD	No Range map
Elagatis bipinnulata	F	DD	No Range map
Enchelycore ramosa	V	H	N/A
Epinephelus daemelii	V	H	N/A
Epinephelus lanceolatus	V	L	N/A
Epinephelus rivulatus	V	H	N/A
Etelis carbunculus	F	DD	Taxonomic confusion
Evistias acutirostris	V	L	N/A
Fistularia commersonii	V	L	N/A
Fistularia petimba	V	L	N/A
Forcipiger flavissimus	V	L	N/A
Girella cyanea	V	H	N/A
Girella fimbriata	V	DD	No Range map

Gorgasia japonica	V	DD	No Range map
Gymnothorax annasona	V	L	N/A
Gymnothorax berndti	V	L	N/A
Gymnothorax obesus	V	L	N/A
Gymnothorax prionodon	V	H	N/A
Hyporthodus octofasciatus	F	L	N/A
Istiophorus platypterus	F	DD	No Range map
Kyphosus bigibbus	V	DD	Taxonomic confusion
Kyphosus sectatrix	V	DD	Taxonomic confusion
Kyphosus vaigiensis	V	DD	Taxonomic confusion
Labracoglossa nitida	V	H	N/A
Lagocephalus lagocephalus	V	DD	Taxonomic confusion
Lagocephalus sceleratus	F	L	N/A
Maxillicosta raoulensis	V	L	N/A
Metavelifer multiradiatus	V	L	N/A
Naso unicornis	V	DD	No Range map
Notolabrus inscriptus	V	H	N/A
Oplegnathus fasciatus	V	L	N/A
Ostracion cubicus	V	H	N/A
Parablennius intermedius	V	H	N/A
Parapercis binivirgata	V	L	N/A
Parioglossus marginalis	V	DD	Taxonomic confusion
Parma kermadecensis	V	H	N/A
Parma polylepis	V	L	N/A
Parupeneus spilurus	V	L	N/A
Platax teira	V	L	N/A
Pomatomus saltatrix	F	DD	Taxonomic confusion

Pseudolabrus luculentus	V	H	N/A
Pterois volitans	V	H	N/A
Scorpaenodes evides	V	DD	Taxonomic confusion
Seriola dumerili	F	L	N/A
Seriola hippos	F	L	N/A
Seriola rivoliana	F	L	N/A
Sphyraena NZ	F	DD	Taxonomic confusion
Sphyraena quenie	F	DD	Taxonomic confusion
Synodus doaki	V	L	N/A
Synodus similis	V	L	N/A
Thalassoma lutescens	V	H	N/A
Thamnaconus analis	V	L	N/A
Trachypoma macracanthus	V	H	N/A
Upeneus francisi	V	L	N/A
Zanclus cornutus	V	H	N/A

Table S3.2 Results for Step 2: species detectability for species assessed by more than one expert panel sub-group. These results were used to highlight any biases or differences in assessments made by the panel sub-groups before panel consensus (note groups still assessed detectability if a species was data deficient from Step 1). 'M' refers to scores for mature life-stages and 'J' refers to juvenile life-stages

		DETECTION	**ABUNDANCE**								**CONSPICUOUSNESS**										**DETECTABILITY**	
Panel Sub-group	**Species**	**Underwater visual (V) or fishing (F) ?**	**How commonly is habitat fished or dived (1 rarely, 2 sometimes, 3 commonly) M**	**How commonly is habitat fished or dived (1 rarely, 2 sometimes, 3 commonly) J**	**Abundance in habitat score M**	**Abundance in habitat score J**	**Abundance total score M**	**Abundance total score J**	**Abundance classification (High/Low) M**	**Abundance classification (High/Low) J**	**Camouflage J (0/1)**	**Body size J (0/1)**	**Hiding J (0/1)**	**Conspicuousness J (0 to 3)**	**Camouflage M (0/1)**	**Body size M (0/1)**	**Hiding M (0/1)**	**Conspicuousness M (0 to 3)**	**Conspicuousness classification M**	**Conspicuousness classification J**	**Detectability classification M**	**Detectability classification J**
Spear	*Acanthocybium solandri*	F	1	1	1	1	2	2	L	L	0	1	1	2	0	1	1	2	H	H	DD	DD
Spear	*Amphichaetodon howensis*	V	1	1	1	1	2	2	L	L	1	0	0	1	1	0	1	2	H	L	H	L
Spear	*Anampses elegans*	V	3	3	2	1	5	4	H	L	0	0	1	1	1	0	1	2	H	L	H	L
Spear	*Aluterus scriptus*	V	2	1	1	1	3	2	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Arripis xylabion*	F	2	2	1	1	3	3	L	L	1	1	1	3	1	1	1	3	H	H	L	L
Spear	*Aulostomus chinensis*	V	3	3	1	1	4	4	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Cheilodactylus ephippium*	V	3	3	2	1	5	4	H	L	1	1	1	3	1	1	1	3	H	H	H	H
Spear	*Cheilodactylus fuscus*	V	3	3	1	1	4	4	L	L	1	1	1	3	1	1	1	3	H	H	H	H
Spear	*Cheilodactylus nigripes*	V	3	3	2	1	5	4	H	L	1	1	1	3	1	1	1	3	H	H	H	H
Spear	*Coris bulbifrons*	V	3	2	1	1	4	3	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Coris picta*	V	2	2	1	1	3	3	L	L	1	0	1	2	1	0	1	2	H	H	H	H

Spear	*Coryphaena hippurus*	F	1	1	1	1	2	2	L	L	0	1	1	2	0	1	1	2	H	H	L	L
Spear	*Elagatis bipinnulata*	F	1	1	1	1	2	2	L	L	1	1	1	3	1	1	1	3	H	H	DD	DD
Spear	*Epinephelus daemelii*	V	2	1	1	1	3	2	L	L	0	1	0	1	1	0	1	2	H	L	H	L
Spear	*Epinephelus lanceolatus*	V	3	2	1	1	4	3	L	L	0	1	1	2	0	1	1	2	H	H	H	H
Spear	*Girella cyanea*	V	3	3	2	1	5	4	H	L	0	1	1	2	1	1	1	3	H	H	H	H
Spear	*Gymnothorax prionodon*	V	3	3	1	1	4	4	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Kyphosus bigibbus*	V	2	2	1	1	3	3	L	L	0	1	0	1	1	1	1	3	H	L	DD	DD
Spear	*Kyphosus sectatrix*	V	2	1	2	1	4	2	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Kyphosus vaigiensis*	V	3	1	1	1	4	2	L	L	0	1	1	2	1	1	1	3	H	H	DD	DD
Spear	*Labracoglossa nitida*	V	2	3	1	1	3	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Spear	*Lagocephalus sceleratus*	F	1	1	1	1	2	2	L	L	0	1	1	2	0	1	1	2	H	H	L	L
Spear	*Naso unicornis*	V	3	2	1	1	4	3	L	L	1	1	1	3	1	1	1	3	H	H	DD	DD
Spear	*Oplegnathus fasciatus*	V	3	2	1	1	4	3	L	L	1	1	0	2	1	1	1	3	H	H	H	H
Spear	*Platax teira*	V	2	1	1	1	3	2	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Spear	*Pomatomus saltatrix*	F	2	1	1	1	3	2	L	L				0	1	1	1	3	H	L	DD	DD
Spear	*Pseudolabrus luculentus*	V	3	3	3	3	6	6	H	H	0	0	1	1	1	0	1	2	H	L	H	H
Spear	*Seriola dumerili*	F	2	1	1	1	3	2	L	L	1	1	0	2	1	1	1	3	H	H	L	L
Spear	*Seriola hippos*	F	2	1	1	1	3	2	L	L	0	1	1	2	0	1	1	2	H	H	L	L
Spear	*Seriola rivoliana*	F	2	1	2	2	4	3	L	L	0	1	0	1	1	1	1	3	H	L	L	L
Spear	*Thalassoma lutescens*	V	3	3	1	1	4	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Spear	*Thamnaconus analis*	V	2	2	1	1	3	3	L	L	0	1	0	1	0	1	1	2	H	L	H	L
Spear	*Zanclus cornutus*	V	2	3	1	1	3	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H

Scuba	*Amphichaetodon howensis*	V	2	2	2	1	4	3	L	L	1	0	0	1	1	0	1	2	H	L	H	L
Scuba	*Anampses elegans*	V	3	3	2	2	5	5	H	H	0	0	1	1	1	0	1	2	H	L	H	H
Scuba	*Aluterus scriptus*	V	3	2	1	1	4	3	L	L	0	1	0	1	0	1	1	2	H	L	H	L
Scuba	*Cheilodactylus ephippium*	V	3	3	2	1	5	4	H	L	0	1	1	2	1	1	1	3	H	H	H	H
Scuba	*Coris bulbifrons*	V	3	2	1	2	4	4	L	L	0	1	0	1	1	1	1	3	H	L	H	L
Scuba	*Coris picta*	V	2	2	2	1	4	3	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Scuba	*Girella cyanea*	V	3	3	2	2	5	5	H	H	0	1	1	2	1	1	1	3	H	H	H	H
Scuba	*Gymnothorax prionodon*	V	3	3	2	1	5	4	H	L	0	1	0	1	1	1	0	2	H	L	H	L
Scuba	*Labracoglossa nitida*	V	2	3	2	2	4	5	L	H	1	0	1	2	1	0	1	2	H	H	H	H
Scuba	*Platax teira*	V	3	1	1	1	4	2	L	L	0	1	0	1	0	1	1	2	H	L	H	L
Scuba	*Thalassoma lutescens*	V	3	3	1	1	4	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Scuba	*Thamnaconus analis*	V	3	3	1	1	4	4	L	L	0	0	1	1	1	0	1	2	H	L	H	L
Scuba	*Zanclus cornutus*	V	3	3	1	1	4	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Fishing	*Acanthocybium solandri*	F	1	1	1	1	2	2	L	L					0	1	0	1	L	n/a	DD	DD
Fishing	*Arripis xylabion*	F	2	2	2	1	4	3	L	L	1	1	0	2	1	1	0	2	H	H	L	L
Fishing	*Aulostomus chinensis*	V	3	3	1	1	4	4	L	L	0	1	1	2	1	1	1	3	H	H	H	H
Fishing	*Cheilodactylus ephippium*	V	3	3	1	1	4	4	L	L	1	1	1	3	1	0	1	2	H	H	H	H
Fishing	*Cheilodactylus fuscus*	V	3	3	1	1	4	4	L	L	1	0	1	2	1	0	1	2	H	H	H	H
Fishing	*Cheilodactylus nigripes*	V	3	3	1	1	4	4	L	L	1	1	0	2	1	1	1	3	H	H	H	H
Fishing	*Coryphaena hippurus*	F	2	1	2	2	4	3	L	L	0	1	0	1	0	1	1	2	H	L	L	L
Fishing	*Elagatis bipinnulata*	F	2	1	2	1	4	2	L	L	1	1	1	3	1	1	1	3	H	H	DD	DD
Fishing	*Epinephelus daemelii*	V	2	1	1	2	3	3	L	L	0	1	1	2	1	1	1	3	H	H	H	H
Fishing	*Epinephelus lanceolatus*	V	3	2	1	1	4	3	L	L	0	1	1	2	0	1	1	2	H	H	H	H

Fishing	*Kyphosus bigibbus*	V	2	1	1	1	3	2	L	L	0	1	1	2	1	1	1	3	H	H	DD	DD
Fishing	*Kyphosus sectatrix*	V	2	1	1	1	3	2	L	L	0	1	1	2	1	1	1	3	H	H	H	H
Fishing	*Kyphosus vaigiensis*	V	2	1	1	1	3	2	L	L	0	1	1	2	1	1	1	3	H	H	DD	DD
Fishing	*Lagocephalus sceleratus*	F	2	1	2	1	4	2	L	L	0	1	0	1	0	1	1	2	H	L	L	L
Fishing	*Naso unicornis*	V	3	3	1	1	4	4	L	L	1	1	1	3	1	1	1	3	H	H	DD	DD
Fishing	*Oplegnathus fasciatus*	V	3	1	1	1	4	2	L	L	0	0	1	1	0	0	1	1	L	L	L	L
Fishing	*Pomatomus saltatrix*	F	2	2	1	1	3	3	L	L	0	1	1	2	1	1	1	3	H	H	DD	DD
Fishing	*Pseudolabrus luculentus*	V	3	3	3	2	6	5	H	H	1	0	1	2	1	0	1	2	H	H	H	H
Fishing	*Seriola dumerili*	F	2	1	2	1	4	2	L	L	1	1	0	2	1	1	1	3	H	H	L	L
Fishing	*Seriola hippos*	F	2	1	1	1	3	2	L	L	0	1	0	1	1	1	1	3	H	L	L	L
Fishing	*Seriola rivoliana*	F	2	1	2	1	4	2	L	L	0	1	1	2	1	1	1	3	H	H	L	L

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DOCTORATE WITH PUBLICATIONS/MANUSCRIPTS

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4 Recognising signatures of tropicalisation: range dynamics of tropical and subtropical fishes in temperate Aotearoa New Zealand

4.1 *Abstract*

Globally, range shifts of marine species mediated by climate change have resulted in novel species assemblages impacting local biodiversity, ecosystem functioning and commercially important fisheries. However, due to a paucity of monitoring programmes, identifying species range shifts is challenging and is often neglected until the impacts are too large to ignore. Here, we examine occurrences of tropical, subtropical, and rare marine fishes in New Zealand to document their range dynamics, identify those species that are *tohu* (indicators) of change, and identify the areas where biogeographic change is occurring. We use citizen science observations of fishes in New Zealand waters along with a classification method to identify species that are potentially undergoing range shifts and identify taxonomic families that are disproportionately contributing to changes in New Zealand's fish biogeography. We use spatial hotspot analyses to identify areas where different species groups are undergoing range changes, and regions in which there are significantly more species undergoing range changes. Locations with a high occurrence of potentially range shifting fishes were centred in north-eastern New Zealand, with occurrences of potentially range shifting fishes decreasing with increasing latitude. Using our classification method, we found that 6% of focal species occurrences in New Zealand were out-of-range, and 87% of these occurrences represented potential and actual range extensions. The remaining 13% were extralimital vagrants (i.e., solitary juveniles) that may not have endured their first winter in the cooler temperate waters of Aotearoa New Zealand. Out-of-range occurrences were identified in 30 focal families, with Labrids and Pomacentrids disproportionately contributing to out-of-range occurrences. Our results suggest that climate-mediated biodiversity change in New Zealand fishes is occurring, albeit at a slower rate than other regions. Nonetheless, our detection of increased species arrival, and increased persistence

in some species, coupled with predicted future warming, signals that future monitoring of these fishes is required to manage any potential impacts. We present a baseline of the current range of tropical, subtropical, and rare fishes in New Zealand, allowing for the accurate and timely identification of species of interest, areas of change, and to inform potential mitigation of impacts from climate-mediated range shifts.

4.2 *Introduction*

The rate of species re-distribution during the Anthropocene is unprecedented (Chen et al., 2011) and in the marine environment, range shifts are occurring approximately an order of magnitude faster than in terrestrial systems (Poloczanska et al., 2013; Sorte, Williams, & Carlton, 2010). One potential driver for the increased rate of climate-mediated species range shifts in marine ecosystems is that the majority of marine organisms are ectotherms, and accordingly, their distributional range conforms closely to their physiological thermal limits (Heath et al., 2012; Pinsky, Selden, & Kitchel, 2020; Pörtner & Peck, 2010; Sunday, Bates, & Dulvy, 2012). Many marine species are also highly mobile, increasing their ability to avoid locations where environmental conditions have become less suitable and exploit new areas where conditions have become more suitable (Pinsky et al., 2020; Pinsky, Worm, Fogarty, Sarmiento, & Levin, 2013). Such range shifting marine species can impact recipient communities by competing with native species (Arrontes, 2002) and modifying community structure (Holbrook, Schmitt, & Stephens, 1997; Ling, 2008; Madin et al., 2012; Pecl et al., 2017). For example, increased thermal stress and herbivory by range-shifting warmer water invertebrates and fishes have driven a complete regime shift on rocky reefs, from kelp forests to ecosystems dominated by reef-building corals in Japan (Kumagai et al., 2018), and from kelp forests into low-diversity, barrens habitats in Australia (Ling, 2008; Ling et al., 2015; Vergés et al., 2014). In turn, these ecosystem regime shifts have widespread and long-lasting impacts on local fisheries and livelihoods (Sanderson, Ling, Dominguez, & Johnson, 2015), highlighting the need to identify and monitor key species and locations of change before the collapse of the resident ecosystem (Hobday, Cochrane, et al., 2016; Pecl, Hobday, et al., 2014).

The rate of species range shifts and the impacts on the recipient ecosystems vary by species and locality (Chen et al., 2011; Sorte et al., 2010; Sunday et al., 2015; van Gennip et al., 2017, Pinsky et al., 2013; Johnson et al., 2011; Marzloff et al., 2015). Hobday and Pecl (2013) identified 24 regions globally where seawater is warming faster than 90% of the rest of the planet and that also have high rates of species range shifts. The majority of these regions are in the path of boundary currents which have become stronger and extend further toward the poles as a result of climate change. For example, the strengthening of the East Australian Current (EAC) has resulted in some of the fastest increases in ocean temperatures globally (Ridgway, 2007) and the poleward range extensions of 16 intertidal invertebrates into Southeast Australia and 45 fishes which represent approximately 15% of the coastal fish fauna (Last et al., 2011; Pitt, Poloczanska, & Hobday, 2010). Many of these range shifting species are herbivorous, territorial, and competitively dominant and can significantly influence the ecosystem structure of the macroalgal dominated reefs (Ling, 2008; Ling et al., 2015). These hotspots of ocean warming and biodiversity impacts, like south-eastern Australia, provide insight into the causes and consequences of range shifts and can inform strategies to proactively manage the impacts of climate change globally (Hobday et al., 2016; Hobday & Pecl, 2014; Miller, Ota, Sumaila, Cisneros-Montemayor, & Cheung, 2018).

Range shifts have three stages. First, a species needs to immigrate to a new location, then persist in the recipient ecosystem, and eventually breed to establish populations (Bates et al., 2014). These stages of a range shift often involve different life-cycle stages and can be differentially influenced by climate change. For instance, the rate of new species immigration may be influenced by climate-induced changes in the strength and directionality of ocean currents preferentially advecting organisms with high dispersal potential from source populations (Booth, Figueira, Gregson, Brown, & Beretta, 2007). For example, the Gulf Stream is the primary mechanism by which tropical fish larvae are transported seasonally from reefs of the South Atlantic Bight into temperate regions as far north as Long Island,New York (40° N latitude, McBride and Able, 1998; Grothues et al., 2002). In contrast, changes in the local marine environment (e.g., sea surface temperatures, productivity and altered seasonality) can impact the persistence and breeding of new or previously rare species (Bates et al., 2014; Monaco et al., 2020). Broad physiological tolerances, phenotypic plasticity, and ecological generalism can enhance persistence, population increase and reproduction in novel communities (Gilchrist, 1995; Rivadeneira & Fernández, 2005; Bates et al., 2014; Sunday et al., 2015, Monaco et al., 2020). For instance, although obligately corallivorous Chaetodontid fishes (described in Booth, Figueira, Gregson, Brown, & Beretta, 2007) dispersed to south Australia they rarely persisted in these novel communities, however, benthic invertebrate feeders or generalist benthic foraging Chaetodontids persisted and underwent successful range shifts (Feary et al., 2013). Thus, observable marine biodiversity changes may include new species immigrating to a location, increased immigration of certain species, increased survival, maturity, or changes in the seasonal occurrence of species (Poloczanska et al., 2016; Sorte et al., 2010) and different species may do well at different stages of range shifts within a location.

Aotearoa New Zealand (hereafter NZ) is geographically isolated, lacks a contiguous poleward coastline, and is not in direct proximity of a strong boundary current. Whereas contiguous coastlines offer stepping-stone habitats that allow warmer-water species to track isotherms poleward, the closest tropical and subtropical source populations for NZ are eastern Australia, Norfolk Island and Lord Howe Island, all over 1000km away (Francis, Worthington, Saul, & Clements, 1999). It is thought an offshoot of the EAC, the Tasman front, brings warm water, and associated tropical and subtropical fishes, from the tropical Pacific into north-eastern NZ as the East Auckland current (EAUC; Stanton, Sutton, & Chiswell, 1997; Stevens, O'Callaghan, Chiswell, & Hadfield, 2021). The velocity, location and trajectory of the EAUC are highly variable

(Bradford-Grieve, Chang, Manning, Paul, & Tasker, 2005; Shears & Bowen, 2017) and climate-induced changes to coastal ocean temperatures in NZ are spatially decoupled from the influence of this boundary current (Shears & Bowen, 2017). Long-term warming around NZ's coastline has been moderate (~0.1–0.3°C/decade) with the greatest increase in temperatures in Wairarapa - a region not influenced by the EAUC - whereas mean temperatures have increased the least in north-eastern NZ (Sutton & Bowen, 2019). Nonetheless, winter minimum temperatures have increased in north-eastern NZ (0.3°C per decade, Shears & Bowen, 2017), and mean ocean temperatures are expected to increase at a rate of up to 2.5°C century^{-1} (Law et al., 2018).

Tropical and subtropical fishes are among the first range extenders into temperate reef ecosystems (Feary et al., 2013; Sunday et al., 2015). Although tropical and subtropical fishes have been observed in NZ over the last 100 years, NZ is not recognized as a region with high rates of climate-induced range extensions. Globally, a threshold of between 17 and 18°C has been identified as the minimum mean temperature for tropical reef fishes to survive (Booth, Beretta, Brown, & Figueira, 2018; Figueira & Booth, 2010). Present-day winter temperatures in northeastern NZ fall below the 17°C threshold reported for tropical reef fish survival (Philip Sutton & Roemmich, 2001). However, under present ocean-warming predictions, minimum winter temperature in some parts of NZ could surpass thresholds for more cold-tolerant tropical fishes within the next 50 years, resulting in increased persistence of range shifting tropical and subtropical fishes in NZ waters.

NZ's isolation and variability in ocean warming trends suggest that range shifts and biodiversity change in NZ are likely to consist of a mosaic of species arrivals, increased survival of cold-tolerant and generalist species, and changes in the seasonal occurrence and persistence of tropical fishes. Although subtle, these early indicators of biodiversity change in coastal marine environments could forewarn more significant biodiversity impacts. Here, we examine occurrences of tropical, subtropical, and rare marine fishes in NZ to document their range dynamics, identify areas of biogeographic change, and species that could be indicators of long-term change. In NZ, these focal species have previously been identified as potential indicators of climate change impacts on biodiversity (see Chapter 2). First, we classify citizen science observations to identify taxa that are out-of-range according to the method of Middleton et al. (2021) and identify taxonomic families that are disproportionately contributing to range shifts in NZ waters. We anticipate that the tropical and subtropical fishes arriving in in NZ may have undergone range shifts at similar latitudes overseas, but that there may be fewer species owing to NZ’s geographic isolation, and that a different suite of species may be newly colonising (i.e., extralimital vagrants) versus those that are newly persisting (i.e., potential range extensions). Second, we identify geographic hotspots of extralimital vagrancy and range extension. We expected that the immigration and persistence of focal species across NZ’s seascape may vary, revealing regions or locations where certain families or life stages dominate the occurrences. Overall, we present a baseline of the current range dynamics for tropical, subtropical, and rare fishes in NZ, allowing for the accurate and timely identification of species of interest, areas of change, and to inform potential mitigation of these climate change impacts.

4.3 *Methods*

4.3.1 Focal species

We focussed on marine ray-finned fishes (Class Actinopterygii) found in NZ waters (hereafter mainland NZ), excluding the sub-Antarctic islands and the Rangitāhua/Kermadec Islands (Figure 1). We derived a list of specific focal species by searching the two most comprehensive guides to NZ fishes (Roberts et al., 2015 and subsequent updates Roberts et al., 2017, 2019; and Francis, 2012) for all species described using any of the following keywords: vagrant, rare, infrequent, occasional, tropical, subtropical, immigrant, and uncommon. We further refined this focal species list to fishes that could be observed or caught shallower than 50m depth according to Fishbase (https://www.fishbase.org), OBIS (https://obis.org/), or published literature (e.g.,Francis, 2012; Roberts, Stewart, & Struthers, 2015). For migratory species, those that regularly occur in mainland NZ, albeit seasonally, were excluded (e.g., Blue Marlin, *Makaira nigricans*; and Striped Marlin, *Kajikia audax*), but we included rare species (e.g., Mahimahi, *Coryphaena hippurus*; and Wahoo, *Acanthocybium solandri*). This resulted in a list of 151 focal species (Supplement 1, Table S1.1). We assigned all focal species a biogeographic affinity based on their occurrences in OBIS and categories proposed by Roberts et al. (2015): *Widespread*: occurs in the Pacific, Indian and Atlantic Oceans, or in the Pacific and Atlantic Ocean; *Endemic*: limited to NZ; *Australasian:* occurs on any coast of Australia, New Caledonia, The Tasman Sea and NZ; *Indo-Pacific*: occurs in the Indian and Pacific Oceans; *Tropical/subtropical Pacific:* limited to the Pacific Ocean between 40° and -40°. If there were inconsistencies between data sources, we assumed the widest distribution for each species.

4.3.2 Focal species occurrences and classifications

We collated occurrence records of our focal species from peer-reviewed literature, unpublished records held by scientists, underwater observations, photographic records, and other unpublished anecdotal information. Most of the focal species occurrences originated from opportunistic citizen science observations, including observations solicited using the WhatsthatfishNZ Facebook page (www.facebook.com/WhatsthatfishNZ). Data sources varied in type, content, and completeness; hence, we used a two-stage validation system to ensure consistency and reliability of the data accompanying an occurrence.

First, we required that each occurrence was accompanied by some minimum metadata (including location with at least 10km precision, year, and season of the occurrence along with the number of individuals sighted), and a photographic image, so that expert scientists or taxonomists could verify the identity of the focal species. We did not require an image if the occurrence data were sourced from experts directly, including specialist marine scientists or experienced members of the community. For occurrences that only included a location name or description, the authors used Google Earth (http://www.earth.google.com) to determine spatial coordinates, including a measure of coordinate uncertainty. Where possible, observers were asked to provide additional metadata including the sex, life stage, size estimate (or measurement from photo), water depth, depth of occurrence, as well as the habitat where the fish was found.

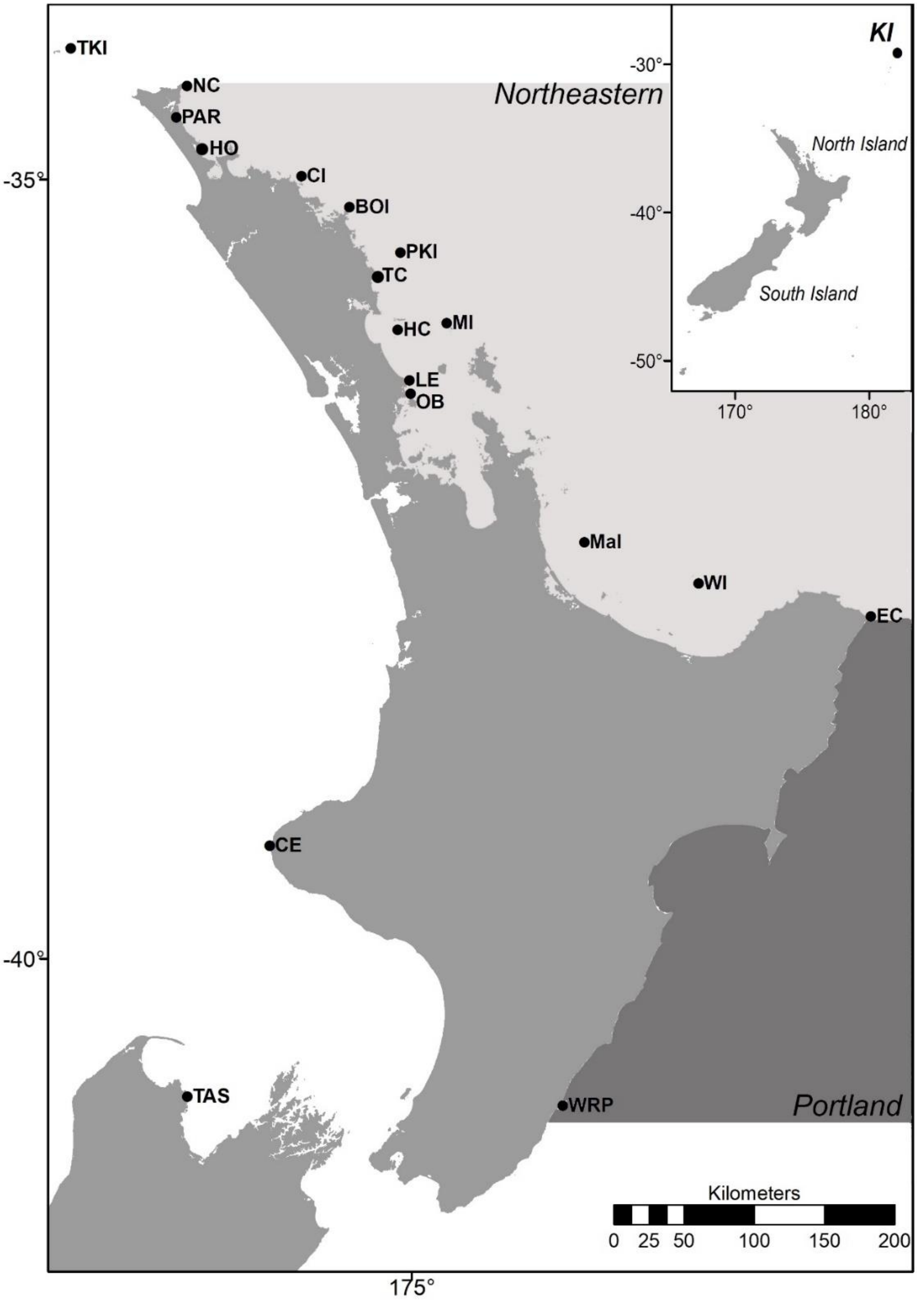

Figure 4.1 New Zealand, including the Northeastern and Portland bioregions as well as the offshore island groups (TKI: Manawatāwhi/Three Kings Islands, CI: Cavalli Islands, PKI: Tawhiti Rahi/Poor Knights Islands, HC: Taranga/Hen and Chicken Islands, MI: Mokohinau Islands, MaI: Tuhua/Mayor Island, WI: Whakaari/White Island) and coastal locations (NC: North Cape, PAR: Parengarenga Harbour, HO: Houhora Harbour, TC: Tutukaka Coast, LE: Leigh, OB: Omaha Bay, EC: East Cape, CE: Cape Egmont, WRP: Wairarapa, TAS: Tasman) mentioned in text.

We used the method of Middleton et al., (2021) to classify focal species occurrences as: within known range, extralimital vagrancy (juvenile individual observed outside of its known range), potential range extension (mature individual observed outside of its known range; or several juvenile individuals observed outside their known range), or a range extension (several mature individuals observed outside of their known range). These classifications, based on maturity and abundance information associated with each species sighting, help to distinguish between the different stages of range extensions (i.e., immigration versus persistence) and thereby potentially the different climate drivers of range dynamics. To apply this method, an expert panel characterised the species known range (i.e., extent of the species' typical geographic distribution) including seasonal variation, if applicable (i.e., extent of the species typical geographic distribution for a given season/s) and assigned a measure of confidence for these defined spatio-temporal range boundaries. Species occurrences were then classified (as described above) based on the observation information and their location relative to the defined spatio-temporal range. An overall confidence score (i.e., no confidence, low confidence, moderate confidence, or high confidence) was assigned to each occurrence classification based on the measure of confidence in the species spatio-temporal range boundaries assigned by the expert panel combined with a species' detectability score. Species detectability was determined using a method similar to that presented in Robinson et al., (2015), based on likely modes of encounter, the species abundance, and conspicuousness (as described in Middleton et al., 2021).

4.3.3 Identification of families most likely to undergo range shifts

Previous studies have identified that some fish families are likely to undergo range shifts before others (Feary et al., 2013). To identify those families with focal species that are more likely to occur outside of their known range in NZ we classified all focal species and native congeners as either out-of-range or within-range. 'Out-of-range' species were defined as those with at least one occurrence that was classified as an extralimital vagrant, potential range extension or range extension. 'Within range' species were identified as NZ native temperate species, focal species for which we do not have occurrences in our database, focal species for which all occurrences were classified as within range using the method of Middleton et al., (2021), or species identified as data deficient due to low confidence in species range or low detectability. We used a binomial generalised linear model with a logit link in fit using the base package in Rstudio v1.1.456 (R Core Team, 2017) to determine if the observed proportion of out-of-range species for a given family was greater or lesser than expected. The response variable was the proportion of out-of-range species for each family with focal species present and we fitted family as a categorical fixed effect with 50 levels. To determine if the proportion of out of range species within a family was greater or lesser than expected, given that data for all fish families with focal species present, we set the intercept as the grand mean proportion of out of range species (ρ=logit[0.2699]) as an offset. We then used one-degree of freedom t-tests to determine if the estimated coefficients for each family were significantly different to the intercept (i.e., the grand mean proportion of out-of-range species)

4.3.4 Spatial patterns of out-of-range occurrences

We mapped spatio-temporal patterns in the number of occurrences for focal species and the distribution of out-of-range focal species occurrences (i.e., extralimital vagrancy, potential range extension or range extension) using ArcMap 10.7.1 (Esri Inc., Redlands, CA). We analysed occurrences by biogeographic affinity (Australasian, Endemic, Indo-Pacific, Tropical and southwestern pacific, and Widespread), classification (extralimital vagrant, potential range extension, or range extension), and taxonomic family. Specifically, for each subset of the focal species records, we applied a 5km square grid across the study area, pooled all occurrences within each grid cell, and assigned it a new common coordinate value using the ‘Spatial join’ tool. We then consolidated all occurrences within that cell to create a new count for the total number of occurrences per grid cell.

To identify grid cells (i.e., locations) that have a greater number of occurrences, relative to other surrounding locations, we conducted 'hotspot' analyses (e.g., Betty et al .2020; Aben et al. 2012; Isobe et al. 2015; Lepers et al. 2005) using the 'Hotspot Analysis' geoprocessing tool in ArcMap 10.7.1 (Esri Inc., Redlands, CA).

As should be expected for citizen-sourced observations, our occurrence locations were not uniformly distributed across the seascape and hence the default number of nearest neighbours (K=8) used in spatial analysis required consideration. We balanced the number of nearest neighbours that would allow us to accurately estimate the spatial relationships and clustering of focal species occurrences, with the number of neighbours that would allow us to make inferences at a sensible spatial scale. While greater numbers of neighbours would provide greater precision, smaller numbers of neighbours allow for more local inferences. To select the value of K for our analysis, we first visualised the relationship between the mean and variance of the Euclidean straight-line distances between our locations at different values of K. Although there was a marked change in the rate of increase of the variance of the mean at a value of K = 8 (Appendix 1, Figure S1.1), the mean spatial distance between locations for K=8 was over just 20km which we considered too large to make local inferences, particularly as many offshore islands have distinctly different fauna and settings but are within 20km of the mainland. Hence, a value of K = 4, which resulted in a mean spatial distance of 12km, was selected as it reduced the effect of noise on the classification and was considered more appropriate given the geographic spread and sparsity of our data in some locations. We used the nearest neighbourhood contiguity scores to construct a spatial weights matrix using the 'Generate Spatial Weights Matrix' tool in ArcMap 10.7.1 (Esri Inc., Redlands, CA) to inform the spatial hotspot analysis.

To identify statistically significant clusters of out-of-range of focal species occurrences in coastal NZ, we calculated the Getis-Ord Gi* statistic (Getis and Ord 1992, 1995) using 'Hotspot

Analysis' geoprocessing tool in ArcMap 10.7.1 (Esri Inc., Redlands, CA). The Gstari* statistic returned a z-score for each location, the larger the z-score is, the more intense the spatial clustering of out-of-range focal species occurrences (i.e., hotspots). Although the hotspot analysis in ArcMap calculates an associated p-value for each z-score, it does so for both hot-spots and cold-spots (i.e., low z-scores or areas with significantly lower concentrations of out-of-range focal species occurrences). Due to the presence-only nature of our dataset we excluded cold spots (negative z-scores) from our analyses and used a one-sided test to determine the significance of the z-scores for each location.

4.3.5 Indicator taxa of change for different regions of New Zealand

Indicator taxa were identified by regional and district jurisdictional boundaries in NZ (according to LINZ Data Service; https://data.linz.govt.nz/). Indicator taxa were taxa that had occurrences classified as an extralimital vagrant, potential range extension or a range extension within that region. We excluded species for which both range confidence and species detectability were 'Low', resulting in 'No Confidence' in the occurrence classification (as per the method of Middleton et al. 2021).

4.4 *Results*

4.4.1 Classification of focal species occurrences

We recorded 1,657 occurrences of 126 focal species from 51 families, including pelagic, reef-dwelling, soft sediment, estuarine and shallow-water species that have varied biogeographic affinities (Supplement 1, Table S1.1). The records spanned more than 100 years with an increase in occurrence data after 1980 (Supplement 2, Table S2.1) and extended from the North Cape in the north of the North Island, to the Tasman region in the north of the South Island (from -42° to -34°; Figure 1). The majority of occurrences were in the Northland (n=1,402), Auckland (n=87) and Bay of Plenty (n=75) regions. We were able to identify the life-stage for 1,011 occurrences, 707 of which were mature fishes and 304 were juvenile fishes. The remainder of the occurrence records (n=646) lacked information on the size or the life-history stage of the individual(s) making maturity difficult to predict. If maturity could not be determined, we excluded these occurrences from further analysis that relied on maturity status.

Most occurrences were within the known range (WR) identified by the expert panel. However, 99 occurrences were identified as out-of-range (OR), including 30 incidences of extralimital vagrancy (EV), 56 potential range extensions (PR), and 13 range extensions (RE) in our study area.

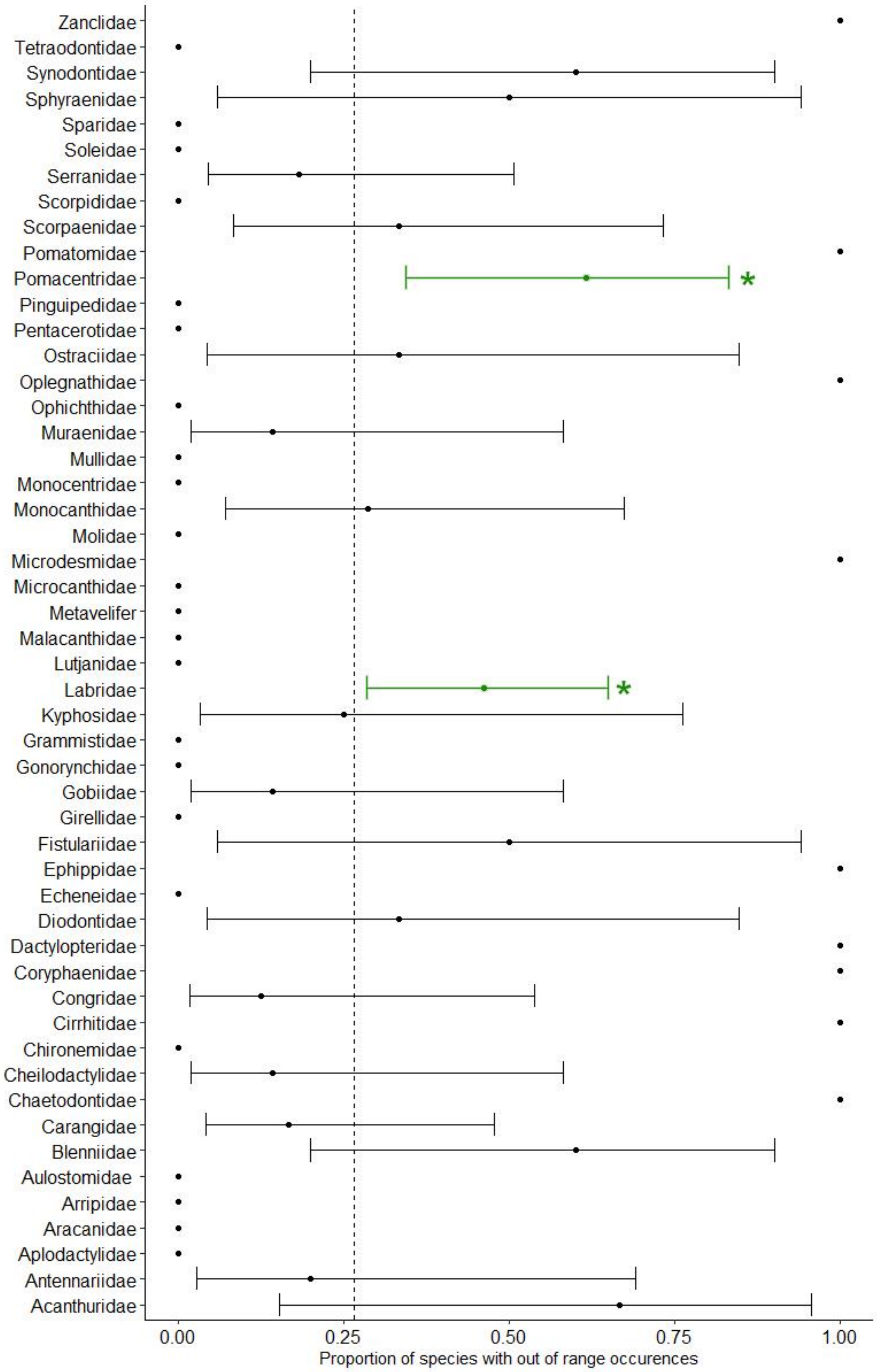

Figure 4.2 The observed proportion of out-of-range species for a given family that contains focal species (± the standard error of predicted mean). The intercept (vertical dashed line) represents the grand mean proportion of out of range species across all families that contain focal species. Lines to the left of the dotted line signify families with fewer out-of-range species than expected; lines to the right of the dotted line signify families that have more out-of-range species than expected. Single points without a line denote families for which no species (zeros) or all species (ones) were out-of-range. Asterisks denote statistically significant differences (p=0.05).

4.4.2 Likelihood of families being out-of-range in New Zealand

Despite a broad range of focal species being identified, out-of-range species were still relatively uncommon (Figure 2). Nine families were represented solely by out-of-range species and 23 families contained only species that had occurrences within their known range. Of the families that contained both out-of-range and within range species, the Acanthuridae, Bleniidae, Labridae, Pomacentridae and Synodontidae families had more out-of-range species than anticipated by chance alone (Figure 2. However, this difference was only significant in the Labrid (Z=2.2; *p*=0.03) and Pomacentrid (Z=2.613; *p*=0.009) families. Many commonly encountered focal families are unlikely to contain out-of-range species, including the Antennariidae, Carangidae, Cheilodactylidae, Congridae, Gobiidae, Muraenidae, and Serranidae.

The number of out-of-range species was highest in the Labridae family, whereas the number of individual out-of-range occurrences was highest in the Pomacentridae family. The majority of extralimital vagrant (EV) occurrences were in the Labridae (n=7) and Pomacentridae (n=9) families (Figure 3). Out-of-range species were most likely to be in the potential range extension (PR) category with 77.2% of PR occurrences due to the presence of single mature fishes, whereas only 22.5% of PRs were due to the observation of multiple juveniles (see Supplement 1, Figure S1.1). Eight species representing eight different families had observations identified as range extensions (RE). The three other families with a high likelihood of having out-of-range sightings (Figure 2) – the Acanthuridae, Blenniidae and Synodontidae – were exclusively represented by RE and PR classifications in NZ (Figure 3).

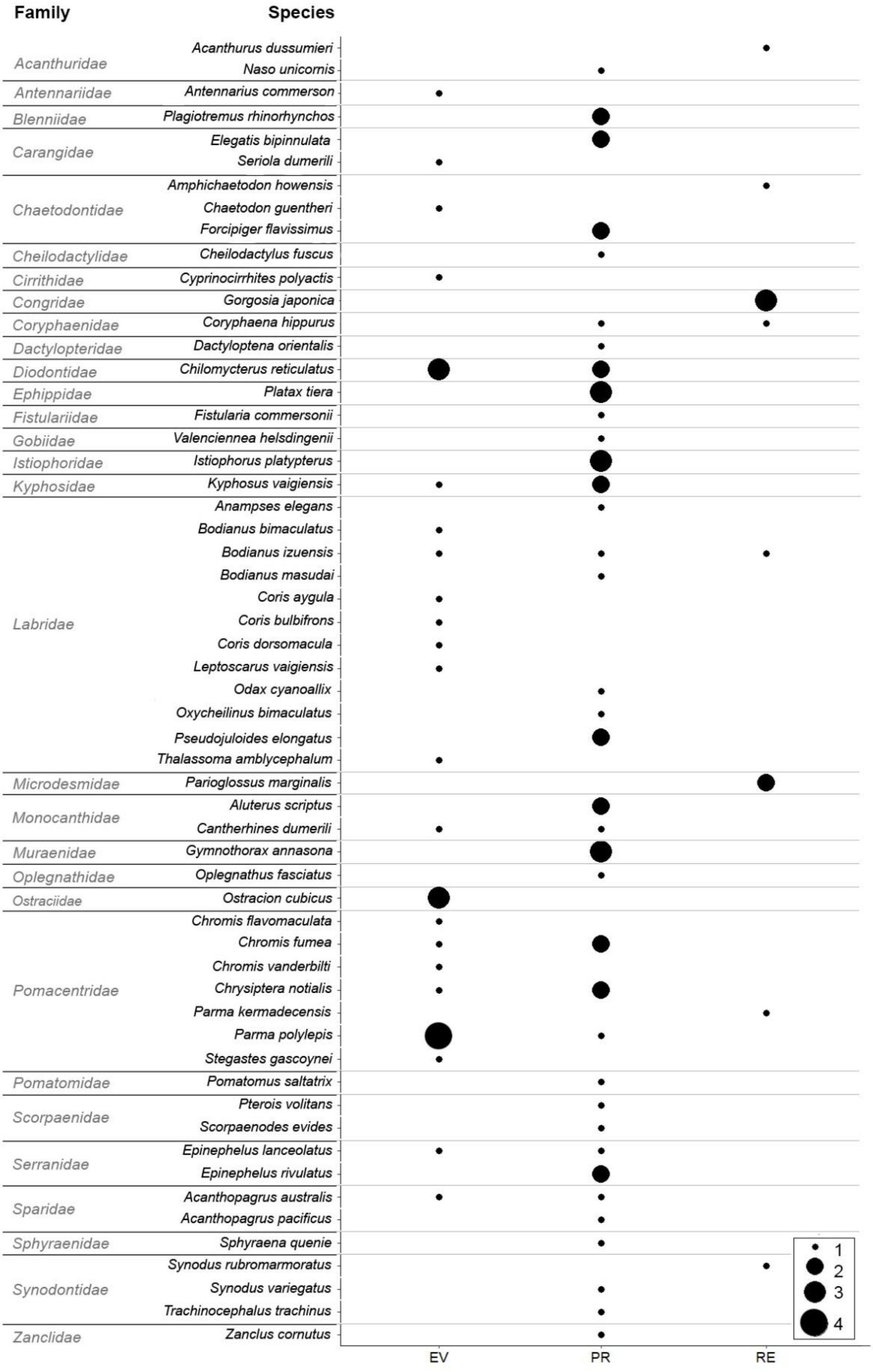

Figure 4.3 Number of out-of-range species occurrences classified as extralimital vagrant (EV), potential range extension (PR) or range extension (RE) using the methods described Middleton et al., (2021).

4.4.3 Geographic hotspots of out-of-range occurrences

Within range occurrences of focal species were spread from the Three Kings Islands in the north to Tasman Bay in the south, with high densities of occurrences around offshore island groups including the Cavalli Islands, Hen and Chickens Islands, Mokohinau Islands, Tuhua/Mayor Island and Whakaari/White Island, along with a number of coastal locations including Parengarenga Harbour, Bay of Islands and Karikari Peninsula (Figure 4).

A statistically significant cluster of within range occurrences was located at the Poor Knights Islands (Z=3.070, *p*=<0.01). The geographic range of the out-of-range observations was similar to within range occurrences but were more diffuse. A significant hotspot of out-of-range occurrences was also identified at the Poor Knights Islands (Z=3.048, *p*=<0.01). When considered individually, occurrences of extralimital vagrant fishes were restricted to areas north of -36° latitude and occurrences were significantly clustered around the Bay of Islands (Z=1.667, *p*=0.05) and Poor Knights Islands (Z=2.580, *p*=<0.01). Hotspots of potential range extensions were also centred around the same two locations Bay of Islands (Z=1.441, *p*=0.07) and the Poor Knights Islands (Z=4.088, *p*=<0.01), but were found further south to -41.3° (Figure4). Distributions of occurrences classified as range extensions were rare (n= 30) and patchily distributed with no significant occurrence hotspots determined.

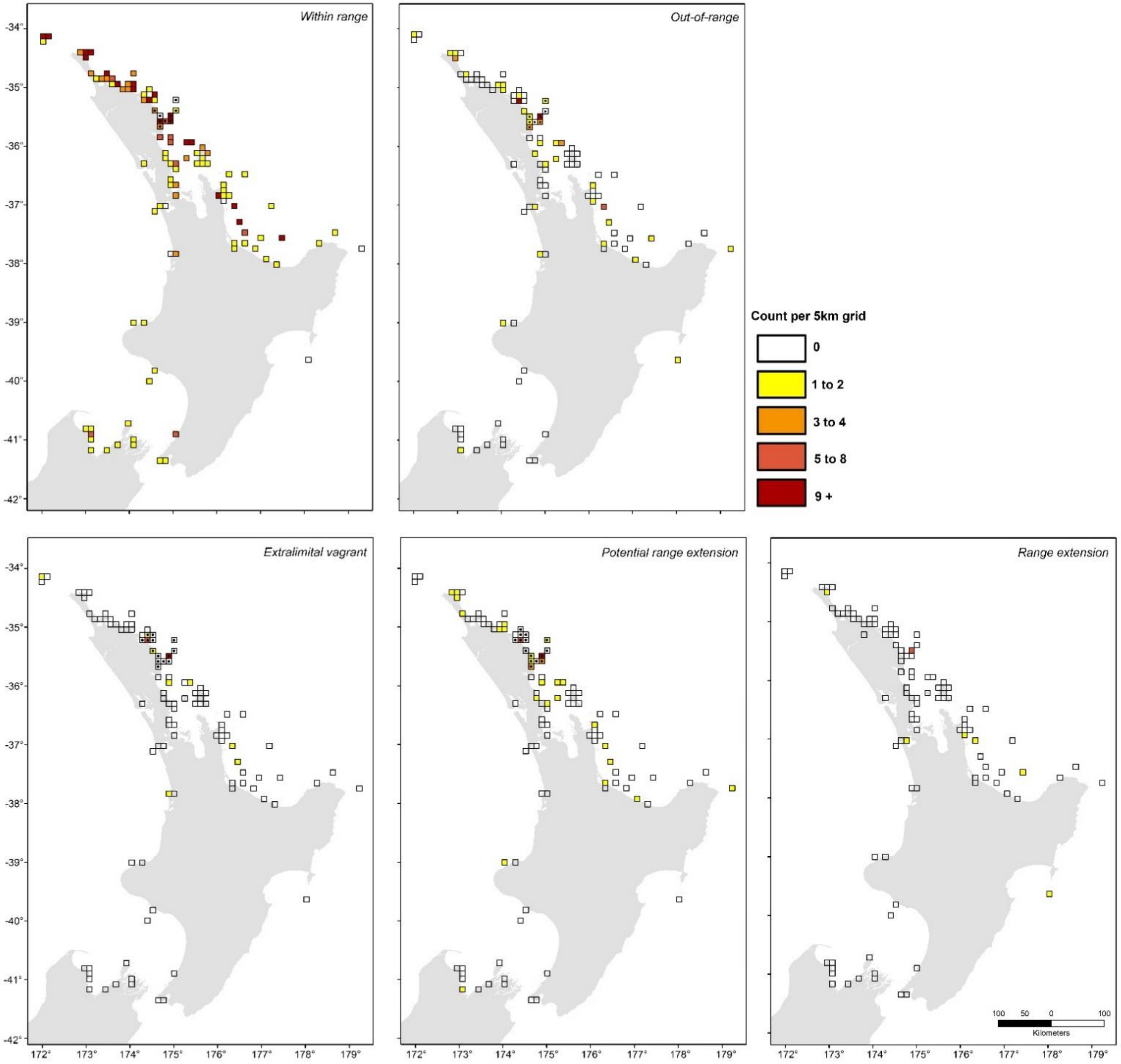

Figure 4.4 Spatial hotspots of focal species occurrences grouped by classification (extralimital vagrant, potential range extension, range extension). Out-of-range occurrences are occurrences from all three classifications combined, within range are all sightings excluding out-of-range sightings. The colour gradient represents the number of occurrences, where darker colours denote locations of high species occurrences. Boxes with a black point are statistically significant (p ≤ 0.10) clusters of high species occurrences for a given classification determined using are the Getis-Ord Gi*statistic and *post hoc* one sided significance tests.

Out-of-range fishes varied in their biogeographic affinity. Out-of-range taxa were identified to have Australasian, Indo-Pacific, Tropical and Southwestern Pacific (hereafter: Tropical SW Pacific), and NZ Endemic biogeographic affinities. Out-of-range occurrences for Indo-Pacific and Tropical SW Pacific species were restricted to north-eastern NZ apart from one occurrence of a Tropical SW Pacific species in Tasman Bay (Figure5). Occurrences classified as out-of-range for Australasian and Widespread species were also located on the west coast of the North Island and south of significant biogeographic boundaries including East Cape and Ahipara (Shears, Smith, Babcock, Duffy, & Villouta, 2008). Australasian, Indo-Pacific and Tropical SW Pacific out-of-range occurrences were significantly clustered at the Poor Knights Islands (Australasian: Z=1.889, *p*=0.03; Indo-Pacific: Z=2.449, *p*=<0.01; and Tropical SW Pacific: Z=2.276, *p*=0.01) the Tutukaka Coast (Australasian: Z=1.889, *p*=0.03; Indo-Pacific: Z=2.449, *p*=<0.01; and Tropical SW Pacific: Z=2.276, *p*=0.01). Out-of-range occurrences of species with Widespread biogeographic affinities were more patchily distributed and hotspots of occurrences were located at the Cavalli Islands (Z=2.630, p=<0.01), Bay of Islands (Z=2.630, p=<0.01), Poor Knights Islands (Z=2.630, p=<0.01), and Tutukaka Coast (Z=2.630, p=<0.01).

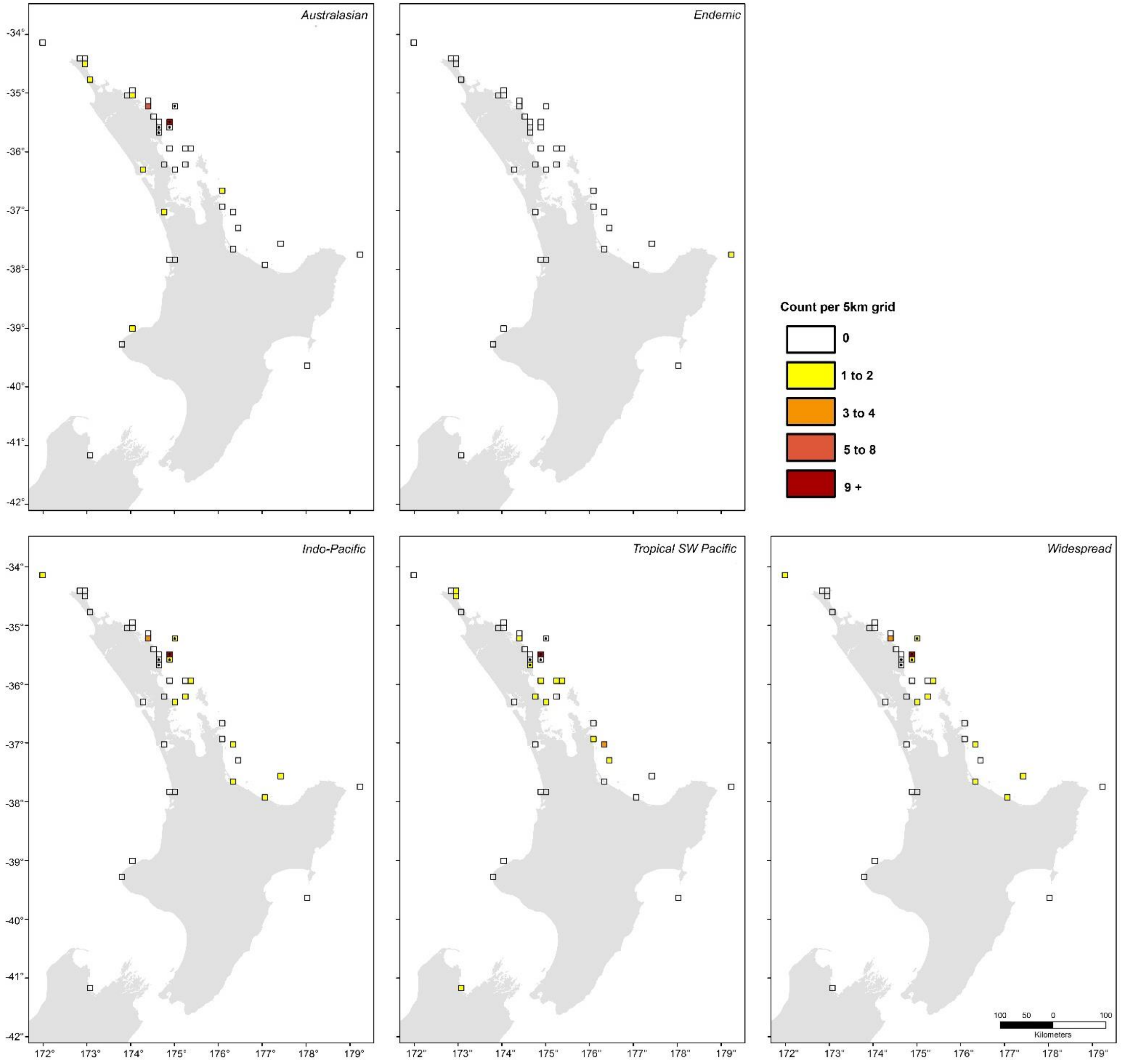

Figure 4.5 Spatial hotspots of out-of-range focal species occurrences grouped by biogeographic affinity (Australasian, NZ Endemic, Indo-Pacific, Tropical and southwestern Pacific, and Widespread). The colour gradient represents the number of occurrences, where darker colours denote locations of high species occurrences. Boxes with a black point are statistically significant ($p \leq 0.10$) clusters of high species occurrences for a given classification determined using the Getis-Ord Gi*statistic and *post hoc* one sided significance tests.

Out-of-range occurrences were recorded for 30 different fish families but were often restricted to one species or observation for many families (Figure 3). Out-of-range occurrences of Blenniidae and Synodontidae species were restricted to areas north of Leigh, whereas occurrences of Acanthuridae, Serranidae and Labridae species extended south to the Bay of Plenty and East Cape (Figure 6). Only four out-of-range occurrences for the most encountered out-of-range families (Acanthuridae, Blenniidae, Labridae, Pomacentridae, Synodontidae and Labridae) were located north of the Bay of Islands, and two south of East Cape (Figure 6). Due to the sparsity of data, hotspots of occurrences could only be identified for Labrids ($Z=2.239$, $p=0.03$) and Pomacentrids ($Z=1.834$, $p=0.07$), both of which had hotspots of occurrences at the Poor Knights Islands (Figure 6).

4.4.4 Indicator taxa of change for different regions of New Zealand

The diversity of focal species that had out-of-range occurrences for which we had low, moderate, or high confidence, was highest in Northland (27 species) and in particular the Whangarei district of Northland (18 species, Figure 7). The most southerly out-of-range occurrences were in the Taranaki and Hawke's Bay regions (Figure 6) with multiple occurrences of Mahimahi (*Coryphaena hippurus*). The diversity of out-of-range species was highest in eastern parts of the North Island and decreased with latitude, with the Taranaki and Auckland regions being the only west coast jurisdictions to have out-of-range occurrences. Notably, a single out-of-range-occurrence for an Endemic NZ fish, the Blue finned butterfish (*Odax cyanoallix),* was identified in the Gisborne region over 600km south of its previously known range. We excluded species for which both range confidence and species detectability were 'Low', resulting in 'No Confidence' in the occurrence classification. This resulted in the retention of 53 out-of-range occurrences of 30 species from 19 families (Figure 7).

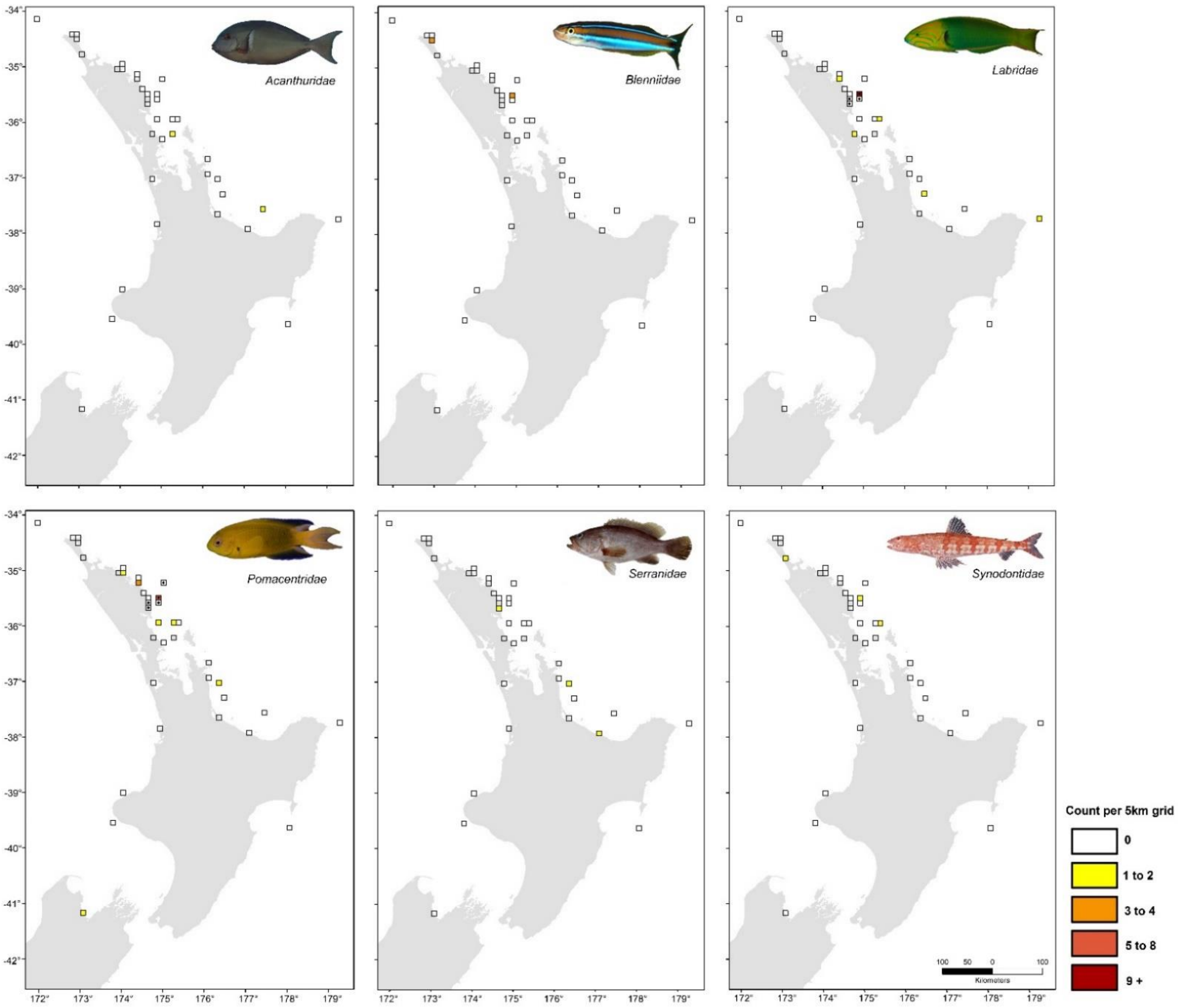

Figure 4.6 Spatial hotspots of out-of-range focal species occurrences of the most common taxonomic families in our database. The colour gradient represents the number of occurrences, where darker colours denote locations of high species occurrences. Boxes with a black point are statistically significant ($p \leq 0.10$) clusters of high species occurrences for a given classification determined using the Getis-Ord Gi*statistic and post hoc one sided significance tests.

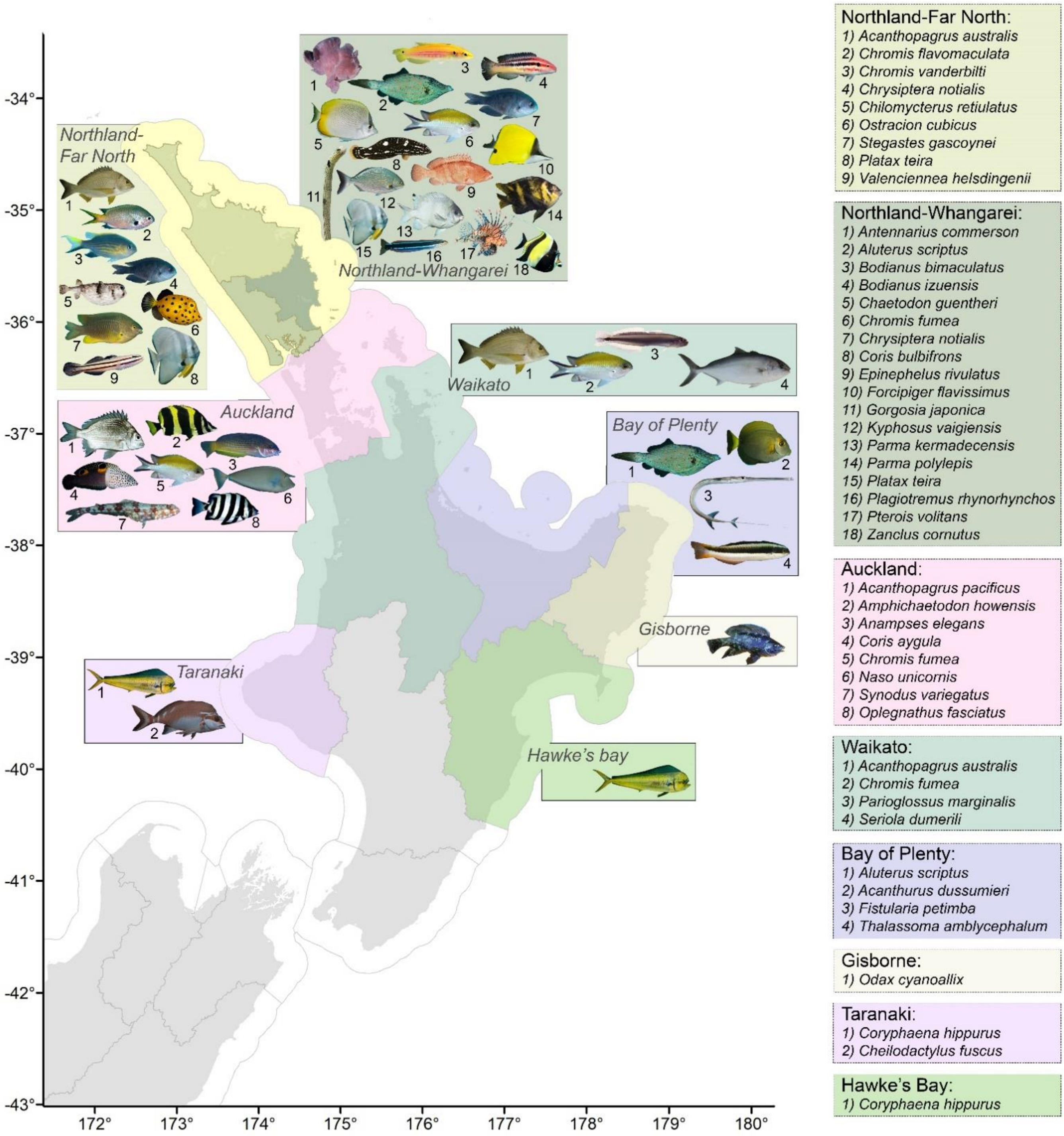

Figure 4.7 Out-of-range species with moderate or high detectability by regional boundaries. Northland had a high number of out-of-range species so the region was split into jurisdictional districts: Whangarei and Far North (the Kaipara district had no occurrences). Colours denote individual districts or regions and the corresponding images of out-of-range species that could be considered 'indicator' species in each region for continued monitoring.

4.5 *Discussion*

The occurrence of tropical fishes in temperate regions is recognised as one of the first indicators of climate-mediated biodiversity change (Munday, Jones, Pratchett, & Williams, 2008; Perry, Low, Ellis, & Reynolds, 2005). Although tropical and subtropical fishes have been recorded in NZ for the past 100 years, very little was known about their distributions, range dynamics and potential impacts. Using citizen science data, and a decision framework guided by expert knowledge, we identify 57 tropical, subtropical and rare fishes that are occurring outside of their historical range in NZ. Of these species, 41 have occurrences that represent potential range shifts and are likely to become a lasting component of NZ's biodiversity. The majority of these potential range shifts occurred in north-eastern NZ – the most biodiverse marine bioregion of mainland NZ (Gordon, Beaumont, MacDiarmid, Robertson, & Ahyong, 2010; Shears et al., 2008). By identifying both the locations of change and those taxonomic groups most likely to be contributing to local biodiversity change, we provide a baseline for future monitoring of species distributions, and knowledge to inform impact mitigation at a local and regional scale. Here, we discuss the patterns revealed for different regions of NZ, the taxonomic families of fishes included in our study, their biogeographic origins, and the potential to include this biodiversity monitoring in the assessment of climate change impacts.

We identified 51 families that were out-of-range in NZ waters (Figure 2); however, only two families, the Pomacentrids and Labrids contributed more out-of-range sightings than we might expect by chance. In comparison, Feary et al. (2013) identified 9-of-55 families that disproportionately contributed to out-of-range occurrences in south-eastern Australia, a climate change hotspot at a similar latitude to NZ. The distance between donor and recipient communities impacts the ecological success of range-shifters (Sorte, Williams, & Carlton, 2010), and it is likely the isolated oceanographic setting of NZ influenced the number, and type (i.e., generalist species with high dispersal capacity and broad physiological tolerances) of species capable of invading NZ. For example, the two most encountered out-of-range families in south-eastern Australia, the Chaetodontidae and Acanthuridae, are found in adjacent coastal areas upstream of temperate recipient locations, whereas these families were underrepresented in our dataset (Figure 2). The suite of out-of-range species we identified in NZ waters reflect the diversity of the closest source populations for tropical and subtropical fishes, namely Norfolk Island, Lord Howe Island and Rangitāhua/Kermadec Islands (Francis et al., 1999). Expressly, the majority of the out-of-range species in our study are of Australasian biogeographic affinities (Figure 5). Therefore, it is likely that NZ's oceanographic setting, the proximity to tropical source populations and species dispersal capacity is reflected in the differences between our results and research from adjacent ocean warming hotspots like south-eastern Australia.

Species that have long pelagic larval duration, high adult mobility and pelagic habitat utilisation or rafting behaviour are most likely to disperse across open ocean habitats (Feary et al., 2013; Luiz et al., 2012). Therefore, we would expect these traits to be common in out-of-range species in isolated island settings like NZ. In our study, the two species with the most out-of-range occurrences *Chilomycterus reticulatus* (n=5) and *Parma polylepis* (n=5), have very different life-history, behavioural and morphological traits. For example, *C. reticulatus* has traits consistent with increased dispersal potential including an extended pelagic phase and the propensity to raft with floating algae (Espino et al., 2019; Robertson, Grove, & McCosker, 2004). Conversely, *Parma* species have benthic eggs, small home ranges and a relatively short pelagic larval dispersal phase, all of which contribute to their low dispersal capacity (Buckle & Booth, 2009; Kingsford & Gillanders, 2000; Tzioumis & Kingsford, 1999). The diversity of out-of-range species and the varied dispersal capacity suggests that mobility alone is not driving the dispersal, and subsequent range shifts, of tropical and subtropical fishes to NZ.

Although the suite of out-of-range species we present reflects those of neighbouring ocean warming hotspots, their ability to persist and undergo range extensions in NZ was highly variable. For example, evidence from proximate ocean warming hotspots suggests that range shifting tropical Labrid and Pomacentrid species have similar cold-water tolerance, and the rate of Pomacentrid arrivals and persistence is higher than those of Labrids (Booth, Beretta, Brown, & Figueira, 2018; Monaco et al., 2020). However, in our study, the majority (60%) of out-of-range Pomacentrid sightings were individual juvenile fish (i.e., extralimital vagrants) and were almost exclusively restricted to warm, northern parts of eastern NZ. In contrast, out-of-range Labrids were distributed as far south as East Cape, and 58% of their occurrences were mature fishes classified as potential range shifts and range extensions. These results suggest that the environment and habitat in NZ waters more closely match the environmental niches of tropical Labrids. Therefore, contrary to the results from previous studies, tropical labrids are more likely to persist long term in temperate NZ waters and undergo range shifts to NZ than out-of-range Pomacentrids. Identifying those novel and out-of-range species that are currently able to persist or those that will persist if warming continues will allow resource managers and researchers to prioritise monitoring and management resources efficiently.

Currently, most tropical, subtropical and rare fish occurrences are in the north-east of NZ. Although the majority of these occurrences are individual juvenile fishes (i.e., extralimital vagrants) or multiple juvenile fishes (potential range extensions), this area is also the only area where mature out-of-range fishes (i.e., range extensions) are clustered (Figure 4). North-eastern NZ is heavily influenced by the warm subtropical EAUC (Brook, 2002; Sutton & Roemmich, 2001) that has been suggested as the most likely oceanographic vector for the transport of tropical and subtropical organisms to NZ (Francis et al., 1999; Shears & Bowen, 2017). Pulses of larval dispersal and adult migration of tropical fishes to NZ occur most summers, but a large proportion of these species consistently undergo significant population declines during winter (Francis et al., 1999). The incidence of mature fishes in north-eastern NZ suggests that this is also one of the few areas where temperatures are currently warm enough to support the persistence and maturity of some tropical and subtropical fishes. Globally, a threshold of between 17 and 18°C is considered the minimum temperature where tropical reef fishes can survive (Booth, Beretta, Brown, & Figueira, 2018; Figueira & Booth, 2010). Although winter temperatures in north-eastern NZ drop to over 2°C below the 17 and 18°C threshold (Philip Sutton & Roemmich, 2001), winter temperatures in north-eastern NZ have increased at rates of between +0.16 and +0.11°C decade^{-1} (Shears & Bowen, 2019). If these trends persist, we will likely observe increased overwintering and long-term persistence of tropical and subtropical fishes arriving in north-eastern NZ.

Evidence suggests that although the oceans around NZ are warming, ocean temperature trends and the subsequent rate of range shifts in NZ are variable. The strongest long-term warming is occurring in areas south of East Cape including the Wairarapa (+0.34°C decade^{-1}) and moderate rates of ocean warming (+0.22°C decade^{-1}) in the Tasman Sea west of NZ (Sutton & Bowen, 2019). Currently, our results suggest that neither the Wairarapa nor the north-western coast of NZ are hotspots for out-of-range species occurrences (Figure 4). Only three out-of-range occurrences were identified in north-western NZ, all of which were species with Australasian biogeographic affinities, and none in the Wairarapa (Figure 5). Conversely, despite identifying north-eastern NZ as a consistent hotspot for out-of-range species sightings this area is not undergoing long-term, annual warming (Sutton & Bowen, 2019).

NZ's marine climate is influenced heavily by the processes occurring in adjacent areas including the Tasman Sea and South-western Pacific. Climate change is predicted to result in the weakening of the Tasman front and strengthening and penetration of the poleward components of the EAC, which could lead to a loss of connectivity between eastern Australia, the Tasman Sea and north-eastern NZ (van Gennip et al., 2017). Due to the changes in the EAC, the Tasman Sea west of NZ is one of the fastest-warming areas in the southern hemisphere. If these changes to the EAC continue, the frequency of out-of-range occurrences in north-eastern NZ, particularly of species that occur in source locations upstream of NZ (including the Australian coast, New Caledonia, and the Tasman Sea), will decrease, but the persistence of any out-of-range species arriving in warming areas like the Wairarapa and those adjacent to the Tasman sea should increase.

The classification method used in this study allowed for the identification of fine-scale spatio-temporal changes in the ranges of several tropical and subtropical fishes in NZ. One example is the Lord Howe Coralfish (*Amphichaetodon howensis*), a species that has been relatively common within their known NZ range since the early 1970's (Supplement 1, Table S1.1). Anecdotal evidence and the absence of juveniles suggests that the population originates from a small number of significant recruitment events in the 1970's and 1980's (M.Francis *pers comm* 2019). Nonetheless, recent observations of juvenile Lord Howe Coralfish at the Poor Knights Islands and Bay of Islands suggest a novel recruitment pulse may have occurred. Conversely, the population of the subtropical Blackspot goatfish (*Parupeneus spilurus*) in NZ is predominantly composed of juvenile fishes. Recent accounts of mature Blackspot goatfish in Northland and the Hauraki Gulf and more recently Whakaari in the Bay of Plenty suggest a shift toward persistence for this species in NZ. Furthermore, several fishes were at different phases of range extension across their NZ ranges. We identified six species (i.e., *Aluterus scriptus, Cantherhines dumerili, Chromis fumea, Chrysiptera notialis, Platax teira and Sphyraena waiti*) that had both potential range extensions and extralimital vagrant occurrences in NZ waters. Such differences across the ranges of these species may be due to complex oceanography influencing dispersal to new locations, or heterogeneous warming of NZ's coastal environment (Shears & Bowen, 2017; Sutton & Bowen, 2019). With continued citizen science monitoring, we anticipate that locations undergoing rapid and long-term warming (i.e., the Wairarapa coast and Southern NZ) might experience an initial increase in sightings of extralimital vagrants followed by observations mature fishes (Shears & Bowen, 2017).

Although many tropical and subtropical fishes are charismatic and conspicuous, they are inherently rare at the leading edges of their range, and during the early stages of establishment.

Long-term, high-resolution monitoring at appropriate scales to detect range shifts is costly, time-consuming, and often spatially restricted (Hortal et al., 2015; Middleton et al., 2021). Citizen science has been useful to overcome a lack of baseline data and detect out-of-range occurrences in both terrestrial (Theobald et al. (2015); e.g., The Great British Birdcount, MonarchWatch.org) and marine settings (Pecl et al. (2019); e.g., Range Extension Database and Mapping Project [Redmap] Reef Environmental Education Foundation [REEF]; European Alien Species Information Network [EASIN]). However, the inferences gained by citizen science may be, in part, determined by the spatial and temporal patterns of citizen activities (Bird et al., 2014) and species conspicuousness (Middleton et al., 2021). Our method is designed to mitigate some of the observational biases in citizen science data by using available scientific data and expert knowledge to determine the known ranges for focal species. Therefore, any out-of-range occurrences at popular and well-studied locations will represent genuine changes in range dynamics (Middleton et al., 2021), particularly in areas like north-eastern NZ where the biogeography of reef fishes is well described (Brook, 2002; Francis, 1996; Francis & Evans, 1992; Francis et al., 1999). For example, our results identified the Poor Knights Islands, a large, old, marine protected area popular with recreational divers, as the most persistent and significant hotspot of out-of-range occurrences (Figure 4). Although it may seem intuitive to conclude that this result is due to observational bias, the out-of-range sightings at the Poor Knights Islands are more likely to represent reliable range shifts due to the broad knowledge of fish diversity at these islands. Nonetheless, observational bias could impact the rate of detection of out-of-range occurrences and future research and data collection effort should focus on locations that are undergoing long-term warming or where data are sparse.

While we have focussed on occurrences and range shifts of tropical and subtropical fishes to NZ waters, these are not the only species impacted by climate change. If climate change mediated ocean warming continues, NZ native temperate fishes are likely to undergo localised range shifts and population changes in response to ambient temperature shifts, habitat loss and novel competition with range shifting species. Our results and anecdotal accounts from citizen scientists suggest that several NZ native fishes are already being sighted outside of their known range, and that there are temporal shifts in seasonal migrations. For example, a spearfisher reported a potential range extension of an endemic Labrid (*Odax cyanoallix*) approximately 400km south of the known southern limit, and since 2016 fishers have reported northern native species such as *Seriola lalandi* and *Chrysophrys auratus* in the far south of NZ (Gibb, 2020; Lewis, 2018). Although our results demonstrate that by utilising novel data sources you can identify species that could be sentinels, or *tohu*, of future change, quantitative data is still limited and disentangling the impacts of other anthropogenic stressors like overfishing and environmental disturbance remains incredibly challenging.

Climate-mediated range extensions do not yet have the same notoriety as invasive species, yet they pose similar risks (Johnson et al., 2011; Sorte et al., 2010). The uncertainty regarding the rate and location of future range shifts and the potential impacts requires a precautionary approach relying on continued monitoring and early intervention. Although regional authorities are mandated and authorised to mitigate local climate change impacts, few have focussed on climate-mediated biodiversity changes. Several existing management tools, utilising relevant legislation such as the Resource Management and Biosecurity Act, could be applied to monitor and mitigate the impacts of range shifting marine species at regional and national scales. Potential management options for regional authorities could include ecosystem management to increase resilience, habitat restoration, species movement or culture restrictions, species-specific control and eradication, and education or awareness (Madin et al., 2012). The proactive application of management tools is recognized as the most effective approach to managing novel species impacts (Giakoumi et al., 2019; Thresher, 1999; Venette et al., 2021). However, a lack of confidence in the likelihood of ecological, socio-economic and management impacts along with an inability to identify range shifting species at early stages of establishment are often cited as justification for not investing resources into range shifting species management.

By developing a method that allows for the prioritisation by species and region in NZ, our study increases the efficiency of monitoring and impact mitigation of range shifting species by resource managers, citizen scientists and researchers. Prioritising highly detectable species should increase the resolution of monitoring data and result in early and efficient identification of range shifts. Furthermore, limiting a focal species list to those that are more likely to persist ensures that resources and monitoring effort is correctly assigned to those species that may have long-term impacts on local biota and ecosystems.

Our data and findings have provided evidence of marine range shifts in a temperate marine setting, identified those species that are likely to contribute to future biodiversity change in the study area and the locations where range shifts are occurring. We hypothesised that the suite of species arriving in NZ will reflect the diversity of the nearest source populations and could be limited by species ability to overcome open ocean dispersal barriers, but that not all species that arrive will persist long term. Indeed, only a small proportion of the out-of-range occurrences were range extensions which suggests that few tropical and subtropical fishes are currently persisting in cooler, temperate waters. The ability to distinguish between ephemeral extralimital vagrant species and those species that can persist in temperate waters allows for efficient identification of those species undergoing range shifts and prioritisation of research and management strategies. Although we demonstrate that tropical and subtropical fishes have undergone range shifts to NZ waters, the spatial distribution of range shifting species in temperate island nations like NZ is likely to be complex. Future research should focus on methods that facilitate the development of reliable and wide-ranging data collection, accurate baselines and early warning systems. As climate change continues to impact marine systems globally, these novel datasets and information sources will facilitate building adaptive capacity of resource managers, scientists, and communities to respond to range shifts proactively with a full complement of applied research and legislative management tools.

4.6 *Authors' Contributions*

IM, LL and JDA conceived and designed the study. IM, developed a focal species database, collected data from citizen scientists, developed quality control measures and collated expert knowledge. LL and IM ran the Whatsthatfish Facebook page to interact with citizen scientists and collect species occurrence data. IM and JDA analysed the data and IM produced the figures. IM drafted the initial manuscript and LL and JDA contributed towards the interpretations of results, the initial drafting and the revisions of the final manuscript.

4.7 *Appendix*

4.7.1 Supplement 1: Full list of focal species and supplementary results

Table S1.1: The full list of focal species for mainland New Zealand including biogeographic affinity and habitat preference. Habitat preference was sourced from Fishbase (Froese, R., and D. Pauly. Editors. 2021.FishBase. World Wide Web electronic publication.www.fishbase.org, (02/2021)). Benthic includes those species found on soft sediment and reef substrates, pelagic are those found in either nearshore or offshore pelagic habitats and Benthic/pelagic are those species that utilise both habitats continuously or during post settlement juvenile and mature phases. Biogeographic affinity was allocated based on definitions from Roberts et al., 2015 (Roberts, C., Stewart, A. L., & Struthers, C. D. (2015). The Fishes of New Zealand (C. D. Roberts, A. L. Stewart, & C. D. Struthers Eds.): Te Papa Press).

Order	**Family**	**Species**	**Biogeographic affinity**	**Habitat preference**
Perciformes	Pomacentridae	*Abudefduf vaigiensis*	Tropical and SW Pacific	Benthic
Perciformes	Serranidae	*Acanthistius cinctus*	Australasian	Benthic
Perciformes	Scombridae	*Acanthocybium solandri*	Widespread	Pelagic
Perciformes	Sparidae	*Acanthopagrus australis*	Tropical and SW Pacific	Benthic
Perciformes	Sparidae	*Acanthopagrus pacificus*	Tropical and SW Pacific	Benthic
Perciformes	Acanthuridae	*Acanthurus dussumieri*	Indo-Pacific	Benthic
Perciformes	Gobiidae	*Acentrogobius pflaumii*	Tropical and SW Pacific	Benthic
Tetraodontiformes	Monacanthidae	*Aluterus monoceros*	Widespread	Benthic/pelagic
Tetraodontiformes	Monacanthidae	*Aluterus scriptus*	Widespread	Benthic
Perciformes	Chaetodontidae	*Amphichaetodon howensis*	Australasian	Benthic

Perciformes	Labridae	*Anampses elegans*	Tropical and SW Pacific	Benthic
Lophiiformes	Antennariidae	*Antennarius commerson*	Indo-Pacific	Benthic
Lophiiformes	Antennariidae	*Antennarius striatus*	Widespread	Benthic
Lophiiformes	Antennariidae	*Antennatus nummifer*	Widespread	Benthic
Perciformes	Aplodactylidae	*Aplodactylus etheridgii*	Widespread	Benthic
Anguilliformes	Ophichthidae	*Apterichthus flavicaudus*	Unknown	Benthic
Perciformes	Gobiidae	*Arenigobius bifrenatus*	Australasian	Benthic
Tetraodontiformes	Tetraodontidae	*Arothron firmamentum*	Widespread	Pelagic
Tetraodontiformes	Tetraodontidae	*Arothron stellatus*	Widespread	Benthic
Perciformes	Arripidae	*Arripis xylabion*	Australasian	Pelagic
Pleuronectiformes	Soleidae	*Aseraggodes bahamondei*	Widespread	Benthic
Perciformes	Microcanthidae	*Atypichthys latus*	Australasian	Benthic
Perciformes	Grammistidae	*Aulacocephalus temmincki*	Indo-Pacific	Benthic
Sygnathiformes	Aulostomidae	*Aulostoma chinensis*	Indo-Pacific	Benthic
Perciformes	Scombridae	*Auxis rochei*	Widespread	Pelagic
Perciformes	Scombridae	*Auxis thazard*	Widespread	Pelagic
Perciformes	Scorpididae	*Bathystethus cultratus*	Australasian	Benthic/pelagic
Perciformes	Labridae	*Bodianus bimaculatus*	Indo-Pacific	Benthic
Perciformes	Labridae	*Bodianus flavifrons*	Australasian	Benthic
Perciformes	Labridae	*Bodianus flavipinnus*	Australasian	Benthic
Perciformes	Labridae	*Bodianus izuensis*	Tropical and SW Pacific	Benthic
Perciformes	Labridae	*Bodianus masudai*	Tropical and SW Pacific	Benthic
Tetraodontiformes	Monacanthidae	*Cantherhines dumerili*	Indo-Pacific	Benthic
Tetraodontiformes	Tetraodontidae	*Canthigaster callisterna*	Australasian	Benthic
Perciformes	Chaetodontidae	*Chaetodon guentheri*	Tropical and SW Pacific	Benthic
Perciformes	Cheilodactylidae	*Cheilodactylus ephippium*	Australasian	Benthic

Perciformes	Cheilodactylidae	*Cheilodactylus fuscus*	Australasian	Benthic
Perciformes	Cheilodactylidae	*Cheilodactylus nigripes*	Australasian	Benthic
Tetraodontiformes	Diodontidae	*Chilomycterus reticulatus*	Widespread	Benthic/pelagic
Perciformes	Pomacentridae	*Chromis abyssicola*	Australasian	Benthic
Perciformes	Pomacentridae	*Chromis flavomaculata*	Tropical and SW Pacific	Benthic
Perciformes	Pomacentridae	*Chromis fumea*	Tropical and SW Pacific	Benthic
Perciformes	Pomacentridae	*Chromis hypsilepis*	Australasian	Benthic
Perciformes	Pomacentridae	*Chromis vanderbilti*	Tropical and SW Pacific	Benthic
Perciformes	Pomacentridae	*Chrysiptera notialis*	Australasian	Benthic
Perciformes	Pomacentridae	*Chrysiptera rapanui*	Australasian	Benthic
Perciformes	Labridae	*Coris aygula*	Indo-Pacific	Benthic
Perciformes	Labridae	*Coris bulbifrons*	Australasian	Benthic
Perciformes	Labridae	*Coris dorsomacula*	Tropical and SW Pacific	Benthic
Perciformes	Labridae	*Coris picta*	Australasian	Benthic
Perciformes	Coryphaenidae	*Coryphaena hippurus*	Widespread	Pelagic
Perciformes	Cirrhitidae	*Cyprinocirrhites polyactis*	Indo-Pacific	Benthic
Scorpaeniformes	Dactylopteridae	*Dactyloptena orientalis*	Widespread	Benthic
Perciformes	Carangidae	*Decapterus muroadsi*	Indo-Pacific	Pelagic
Tetraodontiformes	Diodontidae	*Diodon hystrix*	Widespread	Benthic
Perciformes	Echeneidae	*Echeneis naucrates*	Widespread	Pelagic
Perciformes	Carangidae	*Elagatis bipinnulata*	Widespread	Pelagic
Anguilliformes	Muraenidae	*Enchelycore ramosa*	Tropical and SW Pacific	Benthic
Perciformes	Serranidae	*Epinephelus daemelii*	Australasian	Benthic
Perciformes	Serranidae	*Epinephelus lanceolatus*	Indo-Pacific	Benthic

Perciformes	Serranidae	*Epinephelus rivulatus*	Tropical and SW Pacific	Benthic
Perciformes	Lutjanidae	*Etelis carbunculus*	Unknown	Benthic
Perciformes	Pentacerotidae	*Evistias acutirostris*	Indo-Pacific	Benthic
Sygnathiformes	Fistulariidae	*Fistularia commersonii*	Indo-Pacific	Benthic
Sygnathiformes	Fistulariidae	*Fistularia petimba*	Widespread	Benthic
Perciformes	Chaetodontidae	*Forcipiger flavissimus*	Indo-Pacific	Benthic
Lophiiformes	Antennariidae	*Fowlerichthys scriptissimus*	Indo-Pacific	Benthic
Perciformes	Girellidae	*Girella cyanea*	Australasian	Benthic
Perciformes	Girellidae	*Girella fimbriata*	Endemic	Benthic
Perciformes	Gobiidae	*Gobiopterus semivestitus*	Australasian	Benthic
Gonorynchiformes	Gonorynchidae	*Gonorynchus greyi*	Tropical and SW Pacific	Benthic
Anguilliformes	Congridae	*Gorgasia japonica*	Tropical and SW Pacific	Benthic
Anguilliformes	Muraenidae	*Gymnothorax annasona*	Australasian	Benthic
Anguilliformes	Muraenidae	*Gymnothorax berndti*	Unknown	Benthic
Anguilliformes	Muraenidae	*Gymnothorax obesus*	Australasian	Benthic
Anguilliformes	Muraenidae	*Gymnothorax porphyreus*	Tropical and SW Pacific	Benthic
Anguilliformes	Muraenidae	*Gymnothorax prionodon*	Tropical and SW Pacific	Benthic
Lophiiformes	Antennariidae	*Histrio Histrio*	Widespread	Benthic
Perciformes	Serranidae	*Hypoplectrodes sp.*	Unknown	Benthic
Perciformes	Serranidae	*Hyporthodus octofasciatus*	Widespread	Benthic
Tetraodontiformes	Ostraciidae	*Kentrocapros eco*	Endemic	Pelagic
Perciformes	Kyphosidae	*Kyphosus bigibbus*	Indo-Pacific	Benthic
Perciformes	Kyphosidae	*Kyphosus sectatrix*	Widespread	Benthic
Perciformes	Kyphosidae	*Kyphosus vaigiensis*	Widespread	Benthic
Perciformes	Scorpididae	*Labracoglossa nitida*	Australasian	Benthic/pelagic
Tetraodontiformes	Ostraciidae	*Lactoria diaphana*	Indo-Pacific	Benthic

Tetraodontiformes	Tetraodontidae	*Lagocephalus cheesmanii*	Australasian	Pelagic
Tetraodontiformes	Tetraodontidae	*Lagocephalus lagocephalus*	Widespread	Pelagic
Tetraodontiformes	Tetraodontidae	*Lagocephalus sceleratus*	Indo-Pacific	Benthic
Perciformes	Labridae	*Leptoscarus vaigiensis*	Unknown	Benthic
Perciformes	Malacanthidae	*Malacanthus brevirostris*	Indo-Pacific	
Tetraodontiformes	Molidae	*Masturus lanceolatus*	Widespread	Pelagic
Scorpaeniformes	Scorpaenidae	*Maxillicosta raoulensis*	Australasian	Benthic
Perciformes	Veliferidae	*Metavelifer multiradiatus*	Widespread	Benthic
Tetraodontiformes	Molidae	*Mola tecta*	Unknown	Pelagic
Tetraodontiformes	Monacanthidae	*Monocentris japonica*	Indo-Pacific	Benthic
Perciformes	Acanthuridae	*Naso unicornis*	Indo-Pacific	Benthic
Perciformes	Carangidae	*Naucrates ductor*	Widespread	Pelagic
Tetraodontiformes	Monacanthidae	*Nelusetta ayraud*	Australasian	Pelagic
Perciformes	Labridae	*Notolabrus inscriptus*	Australasian	Benthic
Perciformes	Labridae	*Odax cyanoallix*	Endemic	Benthic
Perciformes	Blenniidae	*Omobranchus anolius*	Australasian	Benthic
Perciformes	Oplegnathidae	*Oplegnathus fasciatus*	Tropical and SW Pacific	Benthic
Tetraodontiformes	Ostraciidae	*Ostracion cubicus*	Indo-Pacific	Benthic
Perciformes	Labridae	*Oxycheilinus bimaculatus*	Indo-Pacific	Benthic
Perciformes	Blenniidae	*Parablennius intermedius*	Tropical and SW Pacific	Benthic
Perciformes	Mullidae	*Parapeneus spilurus*	Indo-Pacific	Benthic
Perciformes	Pinguipedidae	*Parapercis binivirgata*	Unknown	Benthic
Perciformes	Microdesmidae	*Parioglossus marginalis*	Tropical and SW Pacific	Benthic
Perciformes	Pomacentridae	*Parma kermadecensis*	Australasian	Benthic

Perciformes	Pomacentridae	*Parma polylepis*	Australasian	Benthic
Tetraodontiformes	Tetraodontidae	*Pelagocephalus marki*	Widespread	Pelagic
Perciformes	Echeneidae	*Phtheirichthys lineatus*	Widespread	Pelagic
Perciformes	Blenniidae	*Plagiotremus rhinorhynchos*	Indo-Pacific	Benthic
Perciformes	Blenniidae	*Plagiotremus tapeinosoma*	Indo-Pacific	Benthic
Perciformes	Ephippidae	*Platax tiera*	Widespread	Benthic/pelagic
Tetraodontiformes	Ostraciidae	*Polyplacapros tyleri*	Australasian	Benthic
Perciformes	Pomatomidae	*Pomatomus saltatrix*	Widespread	Pelagic
Perciformes	Acanthuridae	*Prionurus maculatus*	Australasian	Benthic
Perciformes	Labridae	*Pseudojuloides elongatus*	Australasian	Benthic
Perciformes	Labridae	*Pseudolabrus luculentus*	Australasian	Benthic
Scorpaeniformes	Scorpaenidae	*Pterois volitans*	Indo-Pacific	Benthic
Anguilliformes	Ophichthidae	*Quassiremus polyclitellum*	Tropical and SW Pacific	Benthic
Tetraodontiformes	Molidae	*Ranzania laevis*	Widespread	Pelagic
Perciformes	Echeneidae	*Remora albescens*	Widespread	Pelagic
Perciformes	Echeneidae	*Remora brachyptera*	Widespread	Pelagic
Perciformes	Echeneidae	*Remora osteochir*	Widespread	Pelagic
Perciformes	Echeneidae	*Remora remora*	Widespread	Pelagic
Perciformes	Scombridae	*Sarda australis*	Australasian	Pelagic
Scorpaeniformes	Scorpaenidae	*Scorpaenodes evides*	Tropical and SW Pacific	Benthic
Perciformes	Carangidae	*Seriola dumerili*	Widespread	Benthic/pelagic
Perciformes	Carangidae	*Seriola hippos*	Australasian	Benthic/pelagic
Perciformes	Carangidae	*Seriola rivoliana*	Widespread	Benthic/pelagic
Perciformes	Sphyraenidae	*Sphyraena acutipinnis*	Indo-Pacific	Pelagic
Perciformes	Sphyraenidae	*Sphyraena qenie*	Widespread	Benthic
Perciformes	Sphyraenidae	*Sphyraena waitii*	Australasian	Pelagic
Perciformes	Pomacentridae	*Stegastes gascoynei*	Australasian	Benthic

Perciformes	Labridae	*Suezichthys arquatus*	Tropical and SW Pacific	Benthic
Perciformes	Labridae	*Suezichthys aylingi*	Australasian	Benthic
Perciformes	Synodontidae	*Synodus doaki*	Tropical and SW Pacific	Benthic
Perciformes	Synodontidae	*Synodus similus*	Tropical and SW Pacific	Benthic
Perciformes	Synodontidae	*Synodus variegatus*	Indo-Pacific	Benthic
Perciformes	Labridae	*Thalassoma amblycephalum*	Tropical and SW Pacific	Benthic
Perciformes	Labridae	*Thalassoma lunare*	Tropical and SW Pacific	Benthic
Perciformes	Labridae	*Thalassoma lutescens*	Tropical and SW Pacific	Benthic
Tetraodontiformes	Monacanthidae	*Thamnaconis analis*	Australasian	Benthic
Perciformes	Synodontidae	*Trachinocephalus trachinus*	Australasian	Benthic
Perciformes	Serranidae	*Trachypoma macracanthus*	Tropical and SW Pacific	Benthic
Perciformes	Mullidae	*Upeneus francisi*	Australasian	Benthic
Perciformes	Gobiidae	*Valenciennea helsdingenii*	Indo-Pacific	Benthic
Perciformes	Zanclidae	*Zanclus cornutus*	Tropical and SW Pacific	Benthic

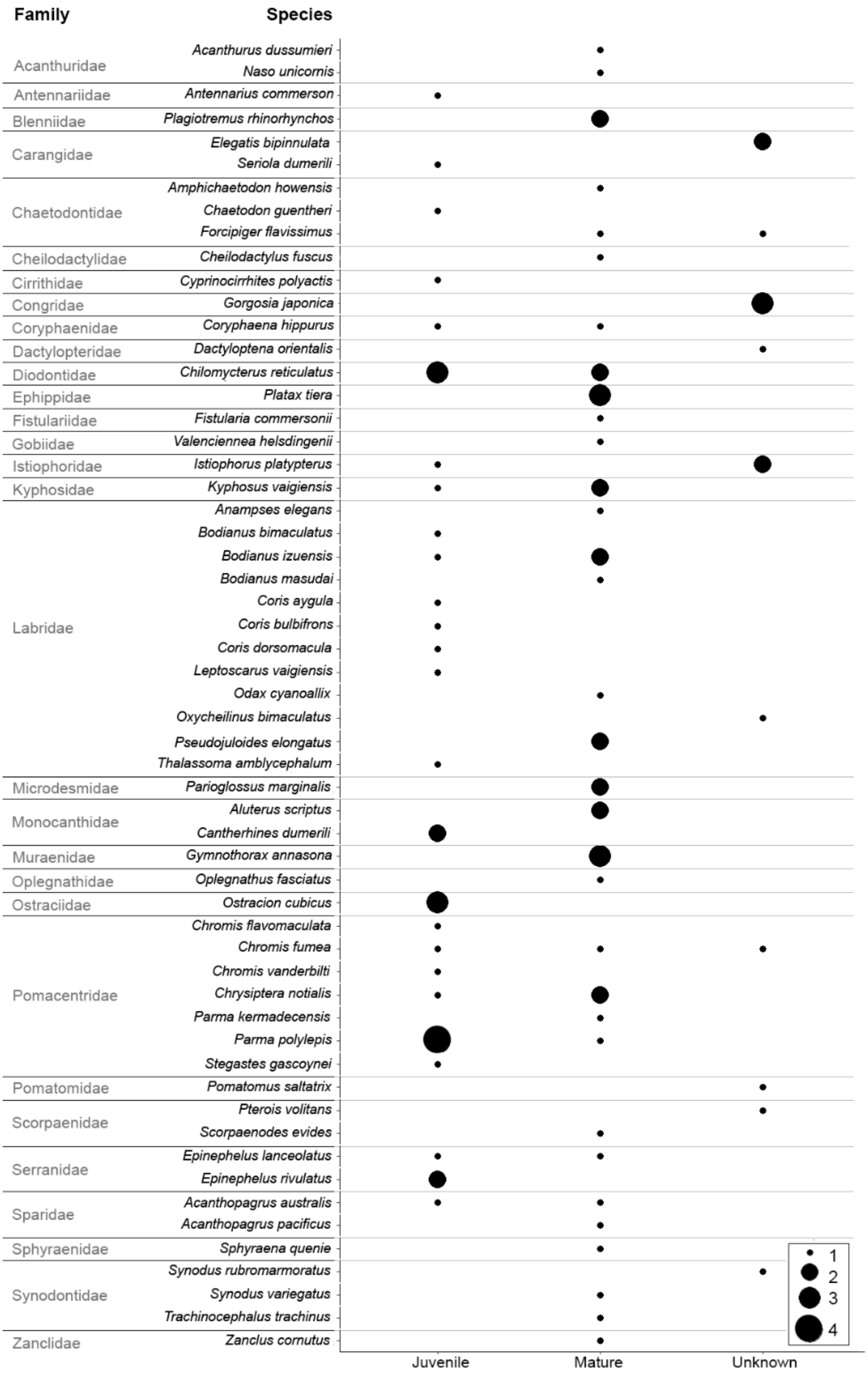

Figure S1.1: The number of out-of-ranges species occurrences classified by life stage.Extralimital vagrant (EV) occurrences are always juveniles, potential range extensions (PR) can be juvenile or mature fishes and range extensions(RE) are always mature fishes.

5 General discussion

5.1 *Major findings and contributions*

There is increasing evidence that climate change is altering the physical and chemical properties of the ocean, which is in turn impacting marine ecosystems and, consequently, the industries and communities that rely on them (Bopp et al., 2013; Pecl et al., 2017; Sweetman et al., 2017; Weatherdon, Ota, Jones, Close, & Cheung, 2016). Climate mediated warming and the associated oceanographic changes are driving regional shifts in abundance and distribution of marine biota (Poloczanska et al., 2013). Shifts in the latitudinal and depth range of species are the most well documented and pervasive ecological impacts of climate mediated warming in marine systems (Burrows et al., 2014; Poloczanska et al., 2013). Globally, poleward range shifts of marine species mediated by climate change have resulted in novel species assemblages that have impacted local biodiversity, ecosystem functioning and commercially important fisheries (Hoegh-Guldberg et al., 2014). However, due to a paucity of monitoring programmes, and the resources required to collect data at appropriate spatio-temporal scales, identifying species range shifts is challenging and often neglected until the impacts are too large to ignore.

Although measurable environmental changes are already occurring (Cummings et al., 2021), examples of climate mediated range shifts and biodiversity change in NZ's oceans are limited. Sea surface temperatures have around NZ have increased by 0.016°C per year since 1981 and are projected to increase by 2.5-3.0°C by 2100 (Law et al., 2018). Based on research in neighbouring regions undergoing similar rates of warming, NZ waters may undergo reductions in primary productivity(Chiswell & Sutton, 2020; Law et al., 2016), losses of temperate water species and increases in subtropical and tropical species (Munday, Jones, Pratchett, & Williams, 2008; Perry, Low, Ellis, & Reynolds, 2005)

This thesis contributes empirical evidence of poleward range shifts and biodiversity change to marine fish fauna within New Zealand waters. I present and test novel methods enabling the use of citizen science observations to identify species contributing to range shifts in NZ and the geographic areas undergoing biodiversity change. Overall, this body of work advances our understanding of NZ's fish fauna, and the potential of citizen science for monitoring climate-mediated biodiversity change in the marine environment. In this chapter I draw together the major findings of the methodological and empirical data chapters of this thesis, discuss the implications of this work for future monitoring and impact management, and propose avenues for future research.

5.2 *Significant contributions*

This thesis provides the first attempt to catalogue rare, subtropical and tropical fishes in NZ waters since 1999 and develop novel tools to allow for the identification of range shifting fishes by combining citizen science and expert knowledge. Furthermore, I identify the species and regions that will be likely to drive future change in NZ waters to streamline monitoring and impact mitigation by scientists and resource managers.

The contributions of **Chapter 2** are twofold; first, I consolidated the current knowledge regarding the contribution of tropical, sub-tropical and extralimital fishes to NZ's marine diversity with the aim to set a current baseline from which to monitor future change. Our results confirm that 25.4% of NZ's teleost fish diversity are tropical, subtropical, and rare fishes (i.e., our focal species) and that the contribution of these fishes to NZ's fish fauna has increased over the last 50 years. Furthermore, I identified 25 families with tropical, subtropical, or rare fishes that contained no NZ native species, and hence represent distinct taxonomic contributions to NZ's teleost fish diversity.

Second, I highlighted the utility of citizen scientists to provide observational data that contributes to our understanding of teleost fish diversity and population dynamics in NZ waters. Scientific monitoring at scales appropriate to detect biodiversity change is costly, time-consuming, and existing data is often spatially restricted. I used citizen science observations to overcome a lack of baseline data and increase the resolution of existing biodiversity records for marine Teleost fishes in NZ waters. Consequently, all I presented 17 new to NZ species records in Chapter 2, all of which were sourced from citizen science observations. These new records represent an increase of the tropical and Southwest Pacific (+0.36%) and Indo-pacific (+0.4%) fishes relative to the last complete assessment of the NZ's fish fauna in 2013 (Roberts, Stewart, & Struthers, 2015). The increase in the contribution of warmer water taxa to NZ's fish diversity suggests that a climate-mediated shift in NZ's fish biodiversity may be underway. I also confidently identified those rare, subtropical and tropical species which are likely to be the first responders to ocean climate change. These early responders could serve as indicator or warning species prior to more broadscale climate change impacts such as biodiversity shifts. Climate driven biodiversity shifts could erode the distinctiveness of this temperate biogeographic region and signal potential future broadscale ecological changes (*sensu* Fogarty, Burrows, Pecl, Robinson, & Poloczanska, 2017)

In **Chapter 3** I develop a novel qualitative method to use citizen science data to determine which species are out-of-range, to distinguish between human mediated species introductions and climate mediated range shifts and identify the stage of colonisation and persistence the species is at. I combined the citizen science data for focal species from Chapter 2 and a decision framework guided by expert knowledge and published research to monitor real-time changes in focal species range dynamics. The method is unique not only in the fact it successfully combines disparate data sources, but also because it classifies individual occurrences rather than species as a whole. This allows the user to continuously monitor species range dynamics, helping to detect real-time, spatially explicit range changes.

I demonstrated the utility of our method by classifying citizen science observations of a subset of 10 focal species from NZ and identified six species with out-of-range occurrences; one of these was likely to be ephemeral, four species had potentially extended their ranges to or within NZ, and one species occurrence was likely due to human-mediated dispersal. Given that there is a paucity of scientific monitoring data in much of the world's oceans, that poleward range shifts are prevalent globally and that the evidence from Chapter 2 suggests that biodiversity shifts are already underway in NZ - novel monitoring approaches are required. Rapid, intuitive, and adaptive methods that allow for the incorporation of disparate data, like those I present in Chapter 3, will become increasingly beneficial biodiversity monitoring and climate change impact management tools.

In Chapter 4, I built on the diversity baseline presented in Chapter 2 using citizen science data and the decision framework presented in Chapter 3 to identify the locations of change and those taxonomic groups most likely to be contributing to local biodiversity change in NZ. I identified 41 tropical, subtropical, and rare fishes that are undergoing range shifts and could become a lasting component of NZ's biodiversity. My results align with research from adjacent ocean warming hotspots (e.g., Feary et al., 2013; Parker et al., 2019; Shalders, Saunders, Bennett, Parker, & Harvey, 2018). Pomacentrid and labrid species were the dominant range shifting taxa and the majority of the potential range shifts occurred in north-eastern NZ – an area adjacent to a major poleward flowing current system that is an offshoot of a Western Boundary Current. Our results also indicate that pelagic species are more likely to present as range shifts in higher latitude locations in NZ than benthic species. Using the results of this chapter I was able to develop a list of range shifting species by jurisdictional boundaries to allow for streamlined management and monitoring by resource managers.

In summary, the empirical chapters in this thesis provide the first analysis of citizen science observations to interrogate the population and range dynamics of tropical, sub-tropical and rare fishes (focal species) in NZ waters in the context of biodiversity shifts. I demonstrate that tropical and subtropical fishes have undergone range shifts to the temperate waters surrounding NZ, and that both sightings and persistence of tropical and subtropical fishes in NZ has increased in the last 100 years. I also illustrate how citizen scientist observations can effectively supplement scientific monitoring of species range dynamics and present a flexible method that facilitates the real-time monitoring of marine range shifts. The updated baseline of fish diversity, methods and focal species lists I present allow for efficient identification of those species currently undergoing range shifts and prioritisation framework that will build the adaptive capacity of resource managers, scientists and communities to respond to range shifts proactively with a full complement of applied research and legislative management tools.

The methods and data presented in this thesis lend themselves to be applied in a variety of settings across a broad range of taxa. Future research should focus on increasing the spatio-temporal resolution of occurrence data for species identified as undergoing range shifts or data deficient in this thesis. Although citizen science observations were a valuable tool for the set of focal taxa in this study, mining alternative datasets, adding *post hoc* statistical analyses or incorporating alternative data collection methods could prove valuable to ongoing monitoring. Furthermore, applying the range shift monitoring methods across a wider geographic area and a broader range of focal species, could help to identify other regions undergoing biodiversity shifts and the local species driving those shifts. Range shifts are likely to continue or increase in NZ waters, and resources or funding are likely always going to limit the extent of mitigation and monitoring projects. Targeted assessments of species impacts, interactions with native or commercially important species, and population dynamics of range shifting taxa will increase the long-term efficiency of management actions taken in response to marine range shifts.

5.3 ***Increased confidence in species ranges and baselines for future monitoring***

An essential prerequisite for my method of identifying range shifts, invasive species and out-of-range occurrences is comprehensive knowledge of species ‘regular’ range. Accurate determination of species ranges usually requires a large amount of information collected over multiple years of standardized fieldwork (Dennis & Thomas, 2000; Mair & Ruete, 2016; Tulloch &

Szabo, 2012). However, the long-term collection of broad-scale information for a wide range of species is often prohibitively expensive (van Strien et al., 2013). I combined novel data sources such as citizen science observations, qualitative data, and published species occurrences with expert knowledge (Chapter 1 & 2), to increase the spatiotemporal resolution of monitoring data sets and determine focal species range boundaries. This approach resulted in increased resolution of the current distributions of tropical, subtropical, and rare fishes in NZ waters, and increased confidence in the range edges for the majority of these species.

Subtropical and tropical fishes in New Zealand are rare (Francis, 2012; Francis, Worthington, Saul, & Clements, 1999) and despite combining various data sources several improvements could be made to increase the accuracy of the baseline distributions for future range shift monitoring. For example, baseline range maps were not able to be developed for any species for which there was taxonomic or range confusion or for which a consensus on range edges. Without a baseline distribution confidence in future range shifts or out-of-range occurrences will be low. Furthermore, the qualitative nature of our method for developing range maps and differences in areas and level of expertise of contributors necessitated a conservative approach. The maps of species ranges are presence only; the experts were confident in identifying where they knew a species was present, but not where it was absent. This led to determination of ranges that may overestimate the area that the fishes occupy. For example, the range for two sub-tropical focal species *Canthigaster callisterna* and *Notolabrus inscriptus* (Figure 5.1)- as determined by the expert panel (Chapter 3)- include the coastline and offshore islands from Cape Maria van Diemen to East Cape. However, published research suggests these species are more common around offshore islands and largely absent from coastal areas (Francis, 2012; Roberts, Stewart, & Struthers, 2015). Overestimation of species distributions could result in fine scale population shifts (i.e., between offshore islands and the mainland) or shifts between habitats within the expert determined range being overlooked. However, any occurrences outside of the expert determined ranges are likely to accurately represent out-of-range occurrences.

The accuracy of the range edges and population structure within the ranges for the focal species could be increased by incorporating additional sources of knowledge and existing data. Local ecological knowledge (LEK, Azzurro, Moschella, & Maynou, 2011; Azzurro et al., 2019), traditional ecological knowledge (TEK, Skroblin et al., 2021) such as Mātauranga Māori (Māori knowledge systems), commercial fisheries data (W. W. L. Cheung, Watson, & Pauly, 2013) and focussed surveys (Edgar & Stuart-Smith, 2014) can increase the accuracy and spatio-temporal scope of the understanding of marine species distributions and range edges. For example, semi-quantitative surveys of small scale commercial fishermen and recreational fishers in the Mediterranean were used to identify ‘new’ species and the northward expansion of warm water fishes in the Mediterranean (Azzurro, Moschella, & Maynou, 2011). Commercial trawl and line fish data has also been used to map the current and historical distribution of all the species currently in the Quota Management System, with the purpose of developing a baseline from which to monitor climate mediated shifts in fish stocks (Dunne, M, *in press*). Although my focal taxa did not include any species that are commercially exploited in NZ fisheries, some species may be caught and recorded as fisheries bycatch and commercial fisheries data could supplement citizen science observations and expert knowledge to increase distributional accuracy in areas not routinely visited by scientists, recreational divers, or fishers.

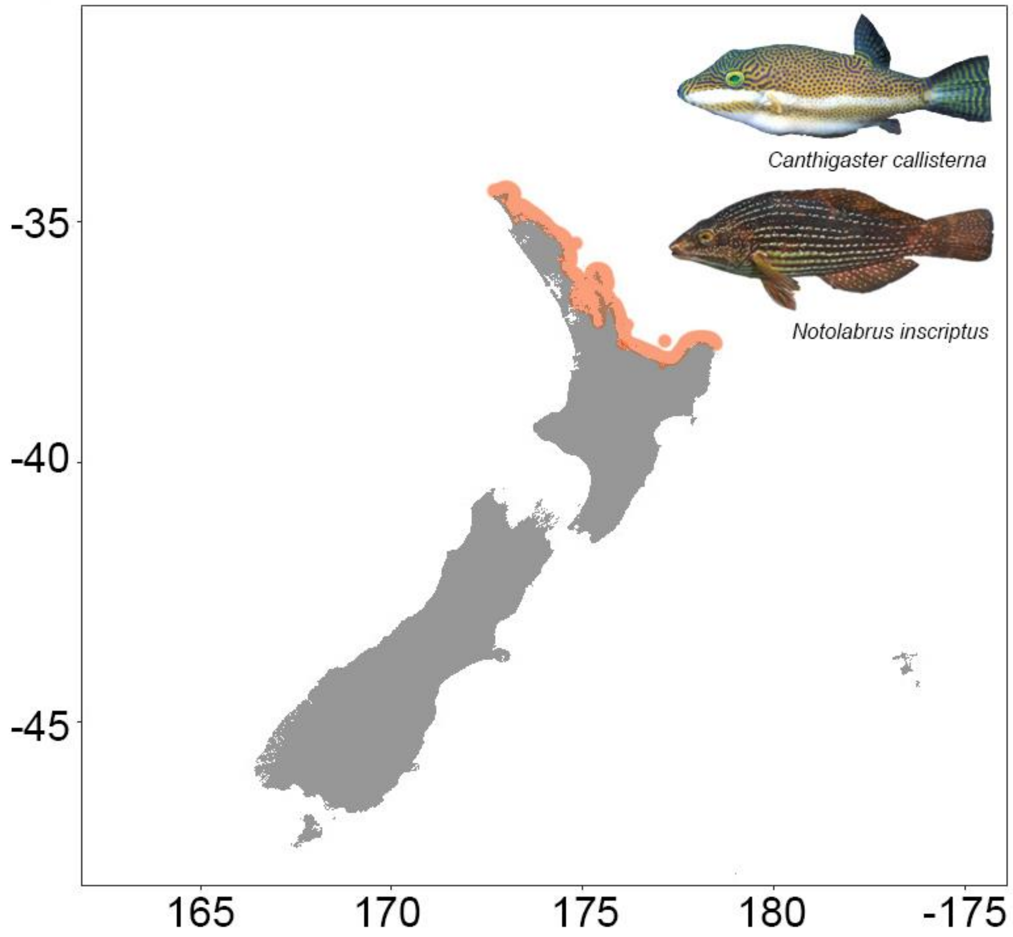

Figure 5.1 Qualitative depiction of the expert determined range (orange) for *Canthigaster callisterna* and *Notolabrus inscriptus*. The range for both species was determined as the entire coastline of north eastern New Zealand from Cape Maria van Diemen in the North to East Cape in the south including all offshore islands within this area.

Traditional ecological knowledge (TEK) including mātauranga Māori (Māori knowledge systems) may be particularly valuable when attempting to determine distributional baselines. Unlike many western science approaches, mātauranga Māori is intimately bound by the fact many generations have resided in one place, and that intergenerational or interfamilial knowledge is transmitted freely (Cheung 2008) This has resulted in individual hapū and iwi (subtribes and tribes) having in-depth localised understanding of the signs and signals of environmental change and the distributions of Taonga species - culturally important species (Paul-Burke, O'Brien, Burke, & Bluett, 2020). The ocean and its resources are of particularly high cultural value to Māori and transdisciplinary research has successfully combined mātauranga Māori with western approaches to inform marine management strategies for species including kina (urchin), *Evechinus chloroticus*; koura (red rock lobster), *Jasus edwardsii*; and pāua (abalone), *Haliotis iris* (Paul-Burke et al., 2020). Although few of the focal taxa in this study are likely to be taonga species, the intergenerationality and temporal extent of mātauranga Māori is likely to be a valuable supplement to scientific distributional data for tropical and subtropical fishes in New Zealand. In fact, informal conversations with northern iwi including Ngāti Kuri, Ngāti Tu and Patuharakeke have suggested that certain focal taxa, such as Mahimahi (*Coryphaena hippurus*) and Northern Kahawai (*Arripis xylabion*) have been occurring in NZ waters for longer than my records suggest and may be more widespread.

My research proves that unstructured and novel data sources, namely – expert knowledge and citizen science observations – can provide baselines from which to monitor distributional change and identify range shifts in the marine environment. Species ranges are dynamic particularly at range edges; and the baselines presented here represent the *current* baselines for the focal species. Climate change and human mediated species introductions will increase the persistence of out-of-range species in novel recipient locations(Monaco et al., 2020; Munday et al., 2008; Sorte, Williams, & Zerebecki, 2010), meaning that ranges will need to be closely monitored and continually redefined. Future research should focus on broadening the knowledge base and data sources used when setting species range boundaries. Furthermore, the use of Species Distribution Models (SDMs), habitat mapping and continued collection of occurrence data could be used to increase the fine- scale resolution of species distributions within their known range.

5.4 *Extending monitoring to a broader range of focal taxa*

Identifying the appropriate sources of data for setting distributional baselines goes hand in hand with identifying the species that are representative of your sampling location and act as indicators of change. In this thesis I use tropical, subtropical, and rare fishes as an indicator of climate mediated range shifts. Globally poleward range shifts of tropical fishes are recognised as early indicators of climate change mediated ocean warming (e.g.,Feary et al., 2013; Parker et al., 2019; Vergés et al., 2014; Wernberg et al., 2011). However, range shifts of temperate species or local taxa may also be informative of climate change mediated regional warming. Although, the geographic ranges of temperate species are often more robust to the impacts of environmental change (Gaston, 1990); anecdotal reports, modelling and fisheries data suggest that NZ native fish distributions and population demographics are responding to climate mediated ocean warming (Dunn et al., *in press*). For example, analysis of fisheries data suggests that the distribution of Snapper (*Chrysophrys auratus*), one of the most popular commercial and recreational fisheries species in NZ, has shifted poleward between 1989 and 2019 (Dunn et al., *in press*). Furthermore, there is evidence of increasing recruitment and catchability towards the southern limit of their range when temperatures are increased (Dunn et al., 2009). Research also suggests that, Kingfish (*Seriola lalandi*) - a popular recreational fishing and spearfishing target species in NZ – are closely track shifting isotherms resulting in shifts in the timing of seasonal migrations and sightings further south than their known range (Champion, Hobday, Zhang, Pecl, & Tracey, 2019; Last et al., 2011).

Anecdotal reports of both snapper and kingfish in southern parts of NZ have increased in recent years (Malcolm Francis *pers comm*), and are often referred to by scientists and citizen observers as ‘sentinels of climate change’ (Lewis, 2018). This statement is supported by research from south-eastern Australia, where poleward range shifts have been identified by citizen scientists for both snapper and kingfish (Champion, Brodie, & Coleman, 2021; Last et al., 2011; Stuart-Smith et al., 2018). My results suggest that the increased arrival and persistence of tropical and subtropical fishes may be a strong signal of climate mediated change in northern NZ. However, poleward extension of native species ranges, like those identified for snapper and kingfish, or the loss of temperate species in northern areas may be some of the observable impacts of climate change at a broader scale across NZ. Therefore, future monitoring of range shifts in NZ should focus not only on the arrival and persistence of novel species but also the population dynamics of native fishes. In particular, the inclusion of recreationally, commercially, and culturally important species will increase the spatial resolution of occurrence records which will in turn allow researchers, resource managers and citizen scientists to better understand the complex patterns of ocean warming in NZ waters and the biodiversity responses to this environmental change.

5.5 *Including more New Zealanders in citizen science monitoring*

The inclusion of recreationally and commercially important fisheries species in future range shift monitoring may facilitate increased engagement of different community groups in underrepresented regions. Tropical and subtropical fishes are highly visible and charismatic, making them ideal for in situ monitoring by divers and spearfishers. However, with the exception of pelagic migratory species such as tuna and mahimahi, tropical and subtropical fishes are not routinely targeted by recreational or commercial fishers. Recreational fishing is one of the most popular maritime activities in NZ, with approximately 30% of all households having at least one recreational fisher (Dodd, Griffiths, le Roux, & Russo, 2020). However, observations made by recreational fishers were underrepresented in my dataset, with the majority of observations being logged by scuba divers and underwater photographers. Scuba diving is not a popular activity in many of the regions undergoing rapid warming (Sutton & Bowen, 2019) and those areas that were underrepresented in my dataset (i.e., Wairarapa, northwestern New Zealand). For example, there are currently no dive shops operating on the East coast of the North Island south of Whakatane and underwater visibility and swell often limit the ability for diving in areas such as the Wairarapa and Taranaki that are hotspots for environmental change (Sutton & Bowen, 2019). Engagement with citizen science projects is often driven by the personal motivation to understand more about species that are important to the contributor. If the primary focus for many ocean-going NZ's is to be able to catch fish as a food source or recreational activity, more thought should be given to selecting the appropriate taxa for each region or sector that will not only identify range shifts and biodiversity change but also lead to maximum rates of citizen scientist engagement.

5.6 *Complementing citizen science data with other data sources and methods*

Monitoring of species distributions and range shifts would ideally employ independent and randomly sampled data with both presence and absence observations. However, in the marine environment such data sets are difficult to obtain, and expensive to collect. Specialist equipment is required to observe marine fishes in-situ and many marine species can be cryptic, highly mobile and have large home ranges. Globally, citizen science has provided a valuable complement to monitoring conducted by government and research organizations (DiBattista et al., 2021). However, the nature of citizen activities and the citizen science data collection methods can result in in spatial and temporal biases in patterns of observations (Robinson, Ruiz-Gutierrez & Fink, 2017). The results presented in this thesis support the value of citizen science observations to detect novel and rare marine species and importantly, our methods were developed to minimise the impacts of observational biases. For example, rather than making broadscale assumptions about the species distributions, we increase spatial resolution and accuracy by classify individual observations as range shifts, vagrancies or within range. This allows for continued revaluation of species distributions and potential shifts of the species range boundaries. Furthermore, our methods are designed to highlight data deficiencies, such as those species or areas for which I need to collect more observations. For example, if observational data is lacking it was unlikely that the expert panel could reach consensus for the baseline distribution of a focal species. Identifying those data deficiencies will allow focussed and more efficient data collection by both citizen scientists and experts alike.

The lack of baseline data for tropical, subtropical and rare fishes in NZ and the restricted timeframe of this study necessitated an *a posteriori* approach to data collection. I mined existing opportunistic citizen science observations from scuba diving, recreational fishing and spearfishing forums and groups and social media. By collating data from diverse and popular sources I created a large data set with minimal additional effort by citizen scientists. However, mining opportunistic records is labour intensive for researchers and expansion to include additional species, locations and data sources is unsustainable long-term. This method of data collection is non-specific and relies on a number of assumptions; first, that the citizen observers are uploading images of all species in all areas at similar rates, second, that the collating scientist is not actively selecting datasources that have limited spatial scope (i.e., local dive stores or forums); and third, that social media, internet and online forums are available and utilised across the entire spatial range of the study. Furthermore, the identification of range shifts for impact mitigation requires timely detection of out-of-range occurrences. Although mining existing opportunistic observation data allows for broadscale data collection at low costs, it may need to be supplemented with targeted data, scientific monitoring or novel data collection methods to detect range shifts in real time.

The data collected, methods, analyses and results presented in this thesis form a framework to design targeted and automated citizen science collection platforms for future monitoring. For example, the results from Chapter 2 set a baseline of focal species distributions and demonstrate that ocean-going citizens are an appropriate focus group for future range shift monitoring. The expert determined species ranges, decision trees, and risk matrices in Chapter 3 were specifically designed to address the observational biases in citizen science programmes and become the framework for an automated web-based citizen science platform to identify out-of-range occurrences of target species (*sensu* RedMap; https://www.redmap.org.au/) and the methods in Chapter 4 can be used to highlight the areas or regions where biodiversity change is occurring to allow for impact mitigation or focussed monitoring. Incorporating the resources presented in this thesis into developing an online contributory science platform is the logical next step to facilitate continued climate-mediated range shift monitoring by citizen scientists and increased resolution of focal species observations in the NZ coastal region.

Although the data collection methods presented here are effective for short term projects like a thesis, it is appropriate to shift to a self-reporting citizen science platform for long term monitoring. Globally, contributory online citizen science platforms have successfully used ‘people power’ to fill knowledge gaps by collecting data in remote marine regions (e.g.,the Secchi Disk Project; http://www.secchidisk.org/), in real-time (e.g.Redmap Australia; https://www.redmap.org.au/) and in high quantities across broad spatial ranges (e.g. the Australasian Fishes Project, https://www.inaturalist.org/projects/australasian-fishes). Furthermore, these contributory projects can act as monitoring programmes (e.g., MPA Watch, https://mpawatch.org/) that directly inform science and conservation policy (DiBattista et al., 2021). Contributory citizen science programmes increase public engagement with the science of ecological change, anthropogenic impacts and species losses (Dickinson et al., 2012; Florence, Jacques, Guillaume, Morgane, & Joséphine, 2020). A user-friendly online platform that allows citizens to upload images of out-of-range species (based on the ranges and data from Chapters 2 & 3), accompanied by a verification pipeline (based on the flow charts from Chapter 3), will facilitate continuous monitoring of marine range shifts in New Zealand and overcome some of the observational biases in citizen science datasets.

Monitoring range shifts and distributional change in the marine environment requires high-resolution observations across a broad spatio-temporal range (Robinson et al., 2020). Citizen science represents an opportunity to collect data at appropriate scales in an efficient cost-effective manner (Gouraguine et al., 2019). As the impacts of anthropogenic climate change increase, identifying the tools and factors that support large-scale citizen science monitoring projects will be increasingly valuable. Recent reviews of the challenges associated with online citizen science platforms, including Redmap (Pecl et al., 2019) and inaturalist (DiBattista et al., 2021), have identified resourcing the development and maintenance of online citizen science platforms, along with citizen engagement and ongoing contribution of data, as the most significant challenges. Additionally, identification of the motivations of citizen scientists, the ease of involvement for participants, and prompt personal feedback are crucial to generating and maintaining ongoing interest by citizens with the project (Pecl et al., 2019). For example, amateur naturalists or underwater photographers may be driven to report novel, rare, or unusual occurrences. Whereas recreational fishers may be more likely to submit images of popular table or sports fish species. Platforms should facilitate reporting of a broad range of species to capture as many target species and occurrence data as possible.

Several considerations need to be taken into account to develop a successful citizen science platform for range shift monitoring in New Zealand’s marine environment. The platform needs to be easy to use, kept up to date, and disseminate the data, research, and science outcomes of the data being contributed back to citizen scientists. Specifically, resourcing will need to cover the set-up costs, promotion of the project, reporting and the ongoing platform or software updates. Furthermore, the focal taxa would need to reflect expert research interests and expertise to ensure long term contributions to species verification pipelines and project refinement. Finally, regionally appropriate taxa would need to be selected to increase engagement with local citizen scientists and the research goals.

Although developing frameworks that facilitate self-reporting will address data deficiencies, resourcing limitations, and spatial biases in citizen science projects, several other approaches are available. For example, research has shown that co-creation citizen science projects - where the public participants and scientists are both are actively involved in the conception, design and execution of the project (Bonney et al., 2009)- often have direct benefits for citizen communities resulting in high levels of public participation and increased longevity (Gunnell, Golumbic, Hayes, & Cooper, 2021; Wiggins & Crowston, 2011). Despite the value of fully co-created projects, they are still rare in ecological monitoring and biological sciences. The majority of citizen science projects are contributory and rely on citizen participation only to collect and analyse data (Kullenberg & Kasperowski, 2016). Most contributory citizen science platforms rely on photographic evidence of occurrences (i.e., inaturalist) or expert verification of records (i.e., RedMap; Wiggins & Crowston, 2011). These approaches increase the quality of citizen derived data but can also impose restrictions on the rate of participation by citizens. Citizen participants will need access to cameras and Wi-Fi to participate, which may be limited in low socio-economic areas that may be most at risk of the social and economic impacts of biodiversity shifts and species losses.

As the number of participants in citizen science increases, variation in skill and commitment levels can lead to decreased precision of the data being collected (Bird et al., 2014). Citizen science coordinators have attempted to maximise the quality of data collected by using expert verification (e.g., Redmap), standardised sampling protocols and training (Edgar & Stuart-Smith, 2014), database management (Crall et al., 2011), and filtering or subsampling data to deal with error and uneven effort (Wiggins & Crowston, 2011, Wiggins et al., 2011). For example, The Reef Life Survey (RLS) project utilises rigorous training and standardised methods to reduce the noise and biases of citizen science data and produced a scientifically robust dataset that quantified densities of fishes, mobile invertebrates, macroalgae and corals at 350 shallow coral reef sites (Edgar & Stuart-Smith, 2014). Data verification and training of citizen scientists is unlikely to eliminate all sources of error and bias. However, *post hoc* statistical analyses can account for the residual error and bias in citizen science datasets. The issues of error and bias found in citizen data are similar to those found in other projects that span broad spatio-temporal scales (Bird et al., 2014). Consequently, existing statistical tools such as generalised linear models (Delaney, Sperling, Adams, & Leung, 2008; Edgar, Barrett, & Morton, 2004), regression splines (Mateo, Felicísimo, & Muñoz, 2010) and hierarchical models (de Solla et al., 2005; Fink et al., 2010; Lonhart, Jeppesen, Beas-Luna, Crooks, & Lorda, 2019) have been used to address identification issues, variability in detection rates and spatial clustering respectively.

As climate change impacts and global redistribution of species continues citizen science datasets like those I present in this thesis will make them increasingly valuable monitoring tools. However, consideration needs to be given to the appropriateness of the collection methods and *post-hoc* analysis to reduce observational biases, increase the scientific rigour of the data, and outcomes of the project, and increase the uptake and application of the data to address conservation and social issues.

5.7 *Predicting range shifts and their impacts.*

Integral to the ability of resource managers and scientists to respond to range shifting species is the ability to detect range shifts at the early stages of establishment and accurately predict the impacts these novel species may have. Range shifts in the marine environment are often diffuse and difficult to distinguish from natural cyclic pulses of population increase and decline (Bates et al., 2014). The methods presented in this thesis will assist in the differentiation of extralimital vagrants and those out-of-range species that can persist in New Zealand waters. However, broadscale and definitive conclusions about the drivers and impacts of range shifts to and within New Zealand are difficult to derive solely from the citizen science dataset I present. Determining how tropical fishes are being transported to New Zealand and the impacts range shifting species may have on recipient communities will require analyses of dispersal capacity, species traits, localised species densities, and the covariates (such as environmental variables or observer bias) that might explain some of the out-of-range observations or perceived range shifts.

Species traits are commonly used to predict which species are likely to disperse to new locations and persist in novel recipient ecosystems (Sunday et al., 2015). Common traits associated with increased range shifts in the marine environment include habitat and dietary generalism, broad physiological tolerance (Bates et al., 2014; Weir & Salice, 2011), rafting behaviour (Luiz et al., 2012), large size at settlement and high mobility (Feary et al., 2013). For isolated island nations like New Zealand, traits associated with dispersal may be particularly relevant predictors of species range shifts. The closest potential sources of subtropical and tropical fish species for mainland New Zealand are the Rangitāhua/Kermadec Archipelago and Norfolk Island, both over 1000km away (Francis, Worthington, Saul, & Clements, 1999). These oceanic distances may serve as dispersal barriers, helping shape the distinct fish assemblages of Norfolk Island, the Rangitāhua/Kermadec Archipelago, and mainland New Zealand (Francis et al., 1999; Roberts et al., 2015). However, some species have physiological, ecological or behavioural traits that improve their likelihood of overcoming such oceanic dispersal barriers (Luiz, Allen, Robertson, Floeter, & Madin, 2015; Luiz et al., 2012). These traits could help predict the likelihood of a species arriving and persisting in an oceanic island nation like New Zealand.

Traditionally pelagic larval duration (PLD) has been viewed as a convenient surrogate for dispersal potential of benthic marine species (Luiz et al., 2013; Luiz et al., 2012). However, recent research suggests that in areas surrounded by significant oceanic barriers that require relatively long transit times, a species propensity to associate (or raft) with marine debris is a better predictor of dispersal potential (Luiz et al., 2012). Marine debris provides an 'oasis' of suitable habitat for benthic species in an otherwise unsuitable environment (Thiel & Gutow, 2005b). It can act as an intermediary settlement substrate, increasing the pelagic dispersal potential of the associated fauna, and can offer shelter or a food source during dispersal across large open stretches (Thiel & Haye, 2006), such as from the tropical and subtropical regions the South Pacific into New Zealand. A species propensity to raft can also increase the establishment rate after arrival in novel recipient communities (Luiz et al., 2015). For example, rafting species have already undergone early life history survival bottlenecks, and only the fittest individuals will arrive in novel recipient communities. Furthermore, marine debris acts as an aggregator for fishes meaning that rafting species are more likely to arrive in groups (Luiz et al., 2012). Both these factors could promote population establishment and persistence in recipient communities.

Marine debris may play a disproportionately significant role in shaping our future marine biodiversity for isolated island nations such as NZ. The increase of persistent marine debris, including plastics, storm debris and fishing waste in our oceans (Thiel & Gutow, 2005a), may facilitate increased dispersal of tropical and subtropical across the oceanic barriers to isolated island nations like New Zealand. Future research should aim to understand better the role of marine debris in the early life history and dispersal phases of species identified as focal or range shifting in this thesis. Furthermore, analyses of the correlation between traits commonly associated with range shifts in ocean warming hotspots (i.e., habitat generalist, large size at settlement, and omnivory) and those species undergoing range shifts in New Zealand should be investigated. When combined with the patterns of range shifts observed in this thesis and ongoing citizen science monitoring, resource managers, scientists and citizens will be better placed to monitor species arrivals and anticipate potential changes to New Zealand's marine fish assemblages and reef ecosystems.

Currently, the methods and data in this thesis provide an opportunity to identify past range shifts and monitor current range shifts as they are happening. However, proactive management of climate-mediated impacts relies on predicting where species may arrive in the future. A natural extension of this study would be to incorporate the citizen science observations with predictive tools such as Species Distribution Models (SDMs). SDM's are a statistical tool widely used in ecological and biogeographic research (Elith & Leathwick, 2009; Kozak, Graham, & Wiens, 2008; Peterson, Papeş, & Soberón, 2015). SDM's correlate species occurrences with environmental predictors and distributional data to determine suitable sites for species (Guisan & Zimmermann, 2000) and predict future climate change impacts on species distributions (Melo-Merino, Reyes-Bonilla, & Lira-Noriega, 2020). Large scale citizen science projects have contributed to the expansion of SDM use over the past twenty years, particularly for biological conservation applications (Guisan & Thuiller, 2005). SDMs rely on two types of data, species distribution data and environmental data. Although large-scale mapping and modelling projects are making high-resolution environmental data is increasingly available (Kozak et al., 2008; Melo-Merino et al., 2020), high-resolution distributional data for many species are often lacking (Scheffers, Joppa, Pimm, & Laurance, 2012). Using large-scale citizen science data to supplement distributional datasets and calibrate or validate SDMs has been proven to increase model accuracy (Robinson et al., 2020) and their application across a broad range of ecosystems.

Opportunistic citizen science datasets like those I present in this thesis can provide a pragmatic solution to data deficiencies in SDMs when quality controlled or filtered for common biases (Yackulic et al., 2013). For example, Robinson et al., (2020) generated a high-quality data set that can closely match structured survey resolution and sampling approach by filtering presence only citizen science data for spatial biases prior to incorporating it into an SDM. To facilitate the use of our citizen science data in SDM modelling to predict the distributions of data deficient species and predict areas where range shifts will occur, citizen scientists should be incentivised to include information related to location and effort when collecting observations. Furthermore, our methods could be extended to include standardised surveys of presence and absences, like those employed by Reef Life Survey (Edgar & Stuart-Smith, 2014), in high-value locations, data deficient areas or areas predicted to undergo significant environmental change (Kelling et al., 2019). These additions will increase the utility of our data to improve the accuracy of SDMs and increase the efficiency and cost-effectiveness of range shift and distributional modelling and surveys (Miller, Pacifici, Sanderlin, & Reich, 2019).

Often range shifting species are assessed together with introduced pest species but given that climate-mediated range extensions do not yet have the same notoriety as invasive species (Johnson et al., 2011; Sorte et al., 2010), they are rarely prioritised for monitoring or impact mitigation. As the rate of climate-mediated species introductions increases, the resourcing of mitigation versus incursion responses and impact management may need to be revised. For instance, legislation regarding ballast discharge, hull fouling, and the aquarium trade (e.g., MPI Import Health Standard: Ornamental Fish and Marine invertebrates 2017) would not have any bearing on climate-mediated range-extending species (Pyke et al., 2008). Instead, a focus on managing species impacts on the recipient ecosystem may need to be prioritised (e.g., culling of the range-extending urchin *Centrostephanus rodgersii* in Tasmania; Sanderson, Ling, Dominguez, & Johnson, 2015).

Although evidence of poleward range shifts of tropical and subtropical fishes is increasing, the impacts of these species in temperate recipient ecosystems are poorly understood. Grazing and reduction of macroalgal cover are the most cited impacts of range shifting tropical species. For example, in Japan (Kumagai et al., 2018), Southern Australia (Wernberg et al., 2016) and the Mediterranean (Sala, Kizilkaya, Yildirim, & Ballesteros, 2011), increases in ocean temperature and a rise in the abundance of tropical herbivorous fishes have coincided with a dramatic decline in canopy-forming macroalgae (Vergés et al., 2014).

Globally, the Labridae family represent one of the most diverse and widespread group of herbivores and habitat modifiers (Choat, Klanten, Van Herwerden, Robertson, & Clements, 2012; Bennett, Wernberg, Harvey, et al., 2015), and they were the most common range shifting taxa in our study (Chapter 4, Figure 4.2). Given that New Zealand reefs are dominated by canopy-forming macroalgae (Shears, Smith, Babcock, Duffy, & Villouta, 2008), the impacts of range shifting herbivorous labrids could be substantial. Focussed citizen science monitoring of the population dynamics of range shifting labrid populations could include repeat monitoring of known populations across seasons, behavioural observations to assess competitive abilities, and dietary or gut content analysis studies. Furthermore, monitoring the resilience of New Zealand macroalgae to ocean warming and grazing by novel herbivores will help disentangle the impacts of climate change on these important ecosystems.

There are no dedicated range shift monitoring programmes in either the terrestrial or marine environment in NZ. However, the public does report some species to the Ministry of Primary Industries as potential pest species, and other opportunistic sightings are incidentally recorded in citizen science platforms or scientific surveys (e.g., inaturalist). New Zealand oceans have warmed at moderate rates (~0.1–0.3°C/decade), and predictions suggest that mean ocean temperatures are expected to increase at a rate of up to 2.5°C century-1 (Law et al., 2018) suggesting that the persistence of tropical and subtropical species in New Zealand will increase. Responsible climate change adaptation of local economies and cultural practices requires an understanding of the potential risks posed by climate change (Adger, Barnett, Brown, Marshall, & O'Brien, 2013; Pecl et al., 2017). The development of dedicated range shift monitoring platforms and the application of the tools presented in this thesis will perform two major functions going forward. First, they will assist in identifying and monitoring climate-mediated species introductions and allow for effective management of potential impacts. Second, they will facilitate increased engagement of the broader community with environmental issues such as climate change.

Given that New Zealand is not experiencing broadscale marine biodiversity shifts in response to climate change and that the monitoring potential of ocean-going citizens in New Zealand is underexploited, we are in a unique position to act proactively. Unlike neighbouring ocean warming hotspots, we have an opportunity to put in place frameworks to monitor biodiversity shifts and minimise the impacts of climate change on our marine environment before they happen. Notably, if scientists and resource managers employ frameworks that include citizen science, they can protect New Zealand's unique marine ecosystem and build relationships with the citizens most likely to be impacted by climate change.

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