Understanding the Population Trends of Shorebirds within New Zealand

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Abstract:

The study investigated population trends of 10 shorebird species in New Zealand, using historic counts from the Ornithological Society of New Zealand dating from 1960-2021. The initial dataset contained counts of 27 shorebird species at 854 local sites within 115 main sites. Minimum population and count cut-offs were created for each species at each main site for winter (June/July) and summer (November/December) census counts, which reduced the final dataset to 10 species (six northern hemisphere-breeding species and four New Zealand-breeding species) at 2–32 sites depending on the season. Generalised Additive Models were used to model count trends over time for each species at each relevant site and season. From these models, population trends for different time periods were extracted: 1990s onwards, 2000s onwards, 2010s onwards, from the first year surveyed and from the year of maximum count. These trends were then used to test if, across the different sites, trends were significantly different to zero (i.e., did trends tend to be negative or positive over each time period). Linear models were then used to test whether site-specific trends varied with local population size (the maximum count), site area (estimated from aerial imagery) and latitude. The aim was to determine if trends varied in a way consistent with a buffer effect, in which population change is greater at low-quality sites (here being sites with smaller populations) than at high-quality sites. Results showed that while trends varied over time for most species at most sites, there was little evidence of a buffer effect. However, trends of northern hemisphere species were more adversely affected than those of internal migrants with 14 time periods of significant decline for the arctic migrants in comparison to 8 time periods of significant decline for the internal migrants.

Acknowledgements:

I would like to thank Phil Battley for his insightful guidance and supervision in the fulfilment of this study which without this research would not have been possible. Thanks to Adrien Riegen and Birds New Zealand for providing access to the count data analysed in this study. Lastly, I would like to thank my parents Paul and Tracey Mowbray for their continued financial and emotional support throughout the course of this project.

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Chapter 1. Introduction:

The order Charadriiformes, commonly known as shorebirds, is an order of birds that consists of roughly 390 species found the world over, in almost all biomes from arctic to tropical (Piersma & Lindström, 2004). Shorebirds are of noted interest to ecologists as due to their globe-spanning habitats, they provide a valuable insight into the state of climate change and its effects on wetland and tidal flat biomes (Mathot et al., 2018; Sutherland et al., 2012), as well as being useful species for assessing the overall health of an ecosystem (Gregory & van Strien, 2010).

Figure 1. Migration Patterns of shorebirds within the East Asia-Australasian Flyway, ranging from Alaska in the north to New Zealand in the South (Szabo et al., 2016)

Study of the important staging sites around the East Asia-Australasian Flyway (EAAF; Figure 1) has shown how detrimental to populations of shorebirds the loss of one site can be, with the continued decline of stopover sites among the Yellow Sea being well documented (Murray & Fuller, 2015; Studds et al., 2017). This has resulted largely from the development and reclamation of intertidal areas for human development on the Chinese and South Korean coast. These areas are of immense importance to shorebird species as studies have shown that they rely on upper intertidal areas (i.e., the most shore-adjacent) for the majority of their foraging (Mu & Wilcove, 2020). These developments have included sites such as Bohai Bay in China, where 450 km^2 of offshore area used by shorebirds was reclaimed for industrial projects over the span of two decades, a loss of a third of the total area (Yang et al., 2011). This is an issue that plagues the entirety of the Yellow Sea coastline, with similar projects being found throughout the region such as at Saemangeum in South Korea, where a sea wall construction in 2006 was shown to cause shorebird numbers at surveyed sites to decrease in by 130,000 birds (Moores et al., 2016). This is not an isolated decline in habitat area though, with studies showing that from the 1950s there has been a noted and steady decline of intertidal areas along the Yellow Sea Coast with a conservative estimate showing a loss of over 730,000 hectares (Murray et al., 2014; Piersma et al., 2016). It is not only the loss of area that has been detrimental to the shorebird population in the Yellow Sea but also the change in quality of sites which remain, with studies showing declines in macrobenthos such as *Potamocorbula laevis*, which acts as one of the primary food sources for shorebirds such as Bar-tailed Godwits, *Limosa lapponica* (Zhang et al., 2018). The intertidal area remaining has been further reduced by the introduction of the invasive plant species, *Spartina*, which since its introduction has spread rapidly claiming areas that previously had been utilised by shorebirds for foraging and roosting (Jackson et al., 2021; Melville et al., 2016). These developments have left shorebirds in the Yellow Sea with no option but to exist in higher density populations. This increased density brings with it more competition for the limited roosting areas and food resources, all while putting populations at a greater risk of disease (Melville et al., 2016). These issues all have compounding effects on the wellbeing of shorebird populations within the Yellow Sea, and as such have major consequences for shorebirds in the EAAF, as virtually all shorebirds using the EAAF pass through the Yellow Sea at least once a year. There are little to no alternative sites and long-term studies show that declines are more prevalent in species that rely on the Yellow Sea area (Sung et al., 2021). This has all resulted in a recorded decline of shorebird numbers by approximately 8% per annum (Studds et al., 2017).

These rather fragile equilibriums between human development and shorebird conservation contain many factors that contribute to the population trends of shorebirds. Aside from the habitat loss there are factors such as pollution from human developments (Egwumah et al., 2017; Tang et al., 2015), and increased predation from introduced species (Dowding & Murphy, 2001) as well as species that previously did not prey on shorebirds but now do due to reduced numbers of preferred prey (McKinnon et al., 2014).

With all these detrimental factors affecting shorebird populations it is of little surprise that the world over there have been recorded declines. In North America, declines in abundance of 70% over 35 years across species have been reported, and an annual decline as large as 15% in Red Knots, *Calidris canutus* (Bart et al., 2007; Ross et al., 2012). Similar declines have been documented in Australia (Clemens et al., 2016). These declines vary in the species and magnitude of the decline, with species such as the Curlew Sandpiper, *Calidris ferruginea,* having a noted 70% decline in Australia since the 1980s (Dawes, 2012) and other shorebirds present such as the Ruddy

Turnstone, *Arenaria interpres,* and Bar-tailed Godwit showing an annual decline of 3% since formal surveys were recorded in the 1970s (Clemens et al., 2016).

New Zealand is ideally located for study of shorebirds, with internal migrant species such as the Wrybill, *Anarhynchus frontalis*, and the South Island Pied Oystercatcher, *Haematopus finschi*, as well as being in the EAAF, with arctic international migrant species such as the Bar-tailed Godwit and Red Knot being prominent species on its shores. This makes New Zealand a particularly interesting location to assess shorebird populations, as both the stressors shorebirds face from migration as well as the threats that face New Zealand's internal migrant species can be seen. New Zealand is home to the largest amount of threatened shorebird species in the EAAF (Milton, 2003)

Previous studies into the populations of shorebirds in New Zealand have largely assessed populations from a national perspective rather than at an individual site level. What these national studies have shown is that populations within New Zealand are declining, with the national populations declining by an average of 1.2% per year since 2005 (Riegen & Sagar, 2020; Sagar et al., 1999). The species that are experiencing declines the most are the arctic migrant shorebirds, an effect likely exacerbated by the continued pressures in the Yellow Sea Stopover region that they pass through on their migrations to New Zealand.

New Zealand internal migrant shorebirds seem to have fared better than arctic migrants, with previous studies showing that most New Zealand shorebird populations have held steady population numbers and some species have shown increases in population numbers thanks to intense conservation efforts, such as the New Zealand Dotterel, *Charadrius obscurus* (Dowding, 2020). Shorebirds within New Zealand face their own stressors with habitat being lost to coastal mangrove growth such as in the Firth of Thames, where some areas that once boasted 40% of the Wrybill population in New Zealand over the winter are now uninhabitable for shorebirds (Woodley, 2005). In the Kaipara Harbour, the intertidal area claimed by mangroves has increased by 41% since the 1960s, while the invasive cord grass *Spartina* has also increased in area (Swales et al., 2013). Environmental factors affect shorebirds in New Zealand with particular regard to New Zealand's internal migrants, such as the Wrybill and South Island Pied Oystercatchers whose habitats of estuarine river mouths are at risk of flooding, with disruptions to their foraging and roosting behaviours (Pierce, 1979;

Riegen & Dowding, 2003). This, coupled with permanent land changes from construction of hydroelectric dams, have left previous ranges inhabited by Wrybill completely unusable, or degraded in both quality and security, with greater access provided to terrestrial predators such as stoats, cats and hedgehogs (Riegen & Dowding, 2003). Changes to the flow of braided rivers that shorebirds inhabit has led to an increase in invasive plant species that reduce the shingle riverbed area that is used for nesting (Riegen & Dowding, 2003). With environmental conditions hampering their preferred habitats they can be faced with new stressors, as they leave their estuaries and river mouths during floods and take refuge on paddocks and sports fields (Sagar & Veitch, 2014); this puts them in closer proximity to both humans and livestock animals, both of which can decrease the breeding success of shorebirds in their presence through disruption of roosting and foraging behaviour to destruction of eggs (Sagar et al., 2000). Other stressors affecting the shorebird populations within New Zealand are much like any habitat near human settlement, predation by introduced pests. In the case of New Zealand studies into the effect of predation by mammals such as cats, rats and mustelids have shown they have a disproportionately large effect on the shorebird populations (Dowding & Murphy, 2001).

New Zealand does have positives for shorebirds such as the New Zealand Dotterel, *Charadrius obscurus*, where the local drivers of population change are well understood. Established conservation management plans have increased breeding success by 28% relative to unmanaged sites (Wills, 2003) and community conservation efforts have led to local population increases in recent years. Projects have also been implemented to revert or compensate for damage done to shorebird habitats, such as Project River Recovery (PRR) in the Mackenzie Basin, which works to restore shorebird habitats in and around braided rivers in the central South Island. The project has used targeted pest control of flora and fauna, as well as creation of wetland habitat, to compensate for damaged habitat from hydroelectric dam construction in the Waitaki Basin (Caruso, 2006). PRR has shown beneficial results for shorebird species, with both Wrybill and Banded Dotterel, *Charadrius bicintus*, having increased nesting and fledging success rate improvements of 27-61% and 71-97%, respectively, since implementation (Rebergen & Woolmore, 2016).

Shorebirds have been censused twice-yearly in Aotearoa/New Zealand for some decades, but the counts have been used largely to estimate total numbers of birds present, and to compare decadal averages (Riegen & Sagar, 2020; Sagar et al., 1999). This approach may say something about current distributions and overall population trends but provides little insight into the role of local factors in changes in numbers, and how equivalent trends may be between sites. This is relevant as it is unlikely that population changes will be experienced equally across sites. A buffer effect can occur where habitats vary in quality, and hence in use by birds (Gill et al., 2001; Gunnarsson et al., 2005). In an expanding population, birds would initially use primarily high-quality sites, but would spread to lower-quality sites as the high-quality sites reached capacity. Conversely, in a declining population, decreases might be experienced first and at higher rates in lower-quality sites, as birds retract their ranges to remain in high-quality sites. In such a situation, high quality habitats could maintain stable populations even when the overall population is declining, or marginal sites could increase proportionately more if high-quality sites are saturated during periods of population increase. In New Zealand, if large intertidal sites (which are typically drowned valleys such as the Kaipara Harbour or enclosed bays such as Farewell Spit) provide better habitat than smaller sites (which are often rivermouth estuaries), a buffer effect may manifest as population changes being emphasised at smaller sites.

My study therefore aims to approach the understanding of shorebird populations in Aotearoa/New Zealand at the individual site level, to answer important questions that cannot accurately be assessed when the populations are viewed on a larger scale. Specifically, I aim to test whether the sites assessed around Aotearoa/New Zealand show a proportional change in numbers of selected shorebird species across all the surveyed sites or whether populations change in a way indicating a potential buffer effect.

The Ornithological Society of New Zealand/Birds New Zealand censuses are described more fully in Chapter 2, but in brief, counts have been made in summer (November/December) and winter (June/July) at as many sites of importance to shorebirds as can be visited by volunteer bird surveyors. A few sites have had continuous survey effort since the 1960s, but most sites only started being counted in the 1980s, and digitised data were available from Birds New Zealand for most sites since the 1990s or 2000s. All shorebirds are counted at each site, but many species of arctic shorebird occur in too small numbers and at too few sites to allow meaningful analysis. Likewise, for New Zealand endemic species we focus on migrants rather than sedentary species.

In this study I will be investigating the population trends of 10 species, Bar-tailed Godwit, Red Knot, Ruddy Turnstone, Pacific Golden Plover (*Pluvialis fulva*), Red-necked Stint (*Calidris ruficollis*), Sharp-tailed Sandpiper (*Calidris acuminata*), South Island Pied Oystercatcher, Pied Stilt (*Himantopus leucocephalus*), Banded Dotterel, and Wrybill.

Bar-tailed Godwit are the most numerous and widespread of the arctic migrants that migrate to New Zealand. The subspecies *baueri* breeds in Alaska and speeds the non-breeding season (and subadult years) New Zealand and Australia; other subspecies breed from Russia to Scandinavia and migrate to Australia across to Africa and Europe. In the 1983-1994 period the average population count of Bar-tailed Godwits present in New Zealand was 83,133 birds (Sagar et al., 1999) which has since fallen to an average count of 77,796 through the 2005-2019 period, a decline of 6.4% (Riegen & Sagar, 2020). However, the initial population estimate for Bar-tailed Godwits in New Zealand was over 100,000 birds, which even on the conservative end of the scale would mean a decline of 22%.

Red Knot are another of the arctic migrants present in New Zealand, migrating from the eastern part of Russia to New Zealand along the EAAF. The population average was estimated to be 58,500 birds during the 1983–1994 period (Sagar et al., 1999) and the average during the 2005–2019 period was 32,080 birds, a decline of 45%. Two subspecies of red knot occur in New Zealand, with the majority being the *rogersi* subspecies that breeds in far eastern Russia on the Chukotka Peninsula, while a minority is the *piersmai* subspecies that breeds on the New Siberian Islands (J. Conklin et al., 2021) The species is known to concentrate on migration through the Yellow Sea in areas with extensive habitat loss (Lok et al., 2019; Rogers et al., 2010; Yang et al., 2011).

Ruddy Turnstone are arctic migrants that are widespread both in breeding habitat and wintering habitat, with an average count of 4,227 birds during the 1983-1994 period (Sagar et al., 1999). Their numbers have decreased dramatically to an average count of 1,654 birds during the 2005–2019 period, a 61% decline (Riegen & Sagar, 2020). This decline is interesting as the Ruddy Turnstone is not as reliant on mudflat for habitats as other species and as such are thought to be less affected by the removal and degradation of habitat that has occurred in the Yellow Sea Stopover area.

Pacific Golden Plover are another arctic migrant present in New Zealand. Their migration from areas of Alaska and Siberia to New Zealand are poorly described, with the first GPS tracking of their migration occurring only in 2019. The population of Pacific Golden Plover has never been large in New Zealand as New Zealand is located at the end of their migration range with much of the population being found in Australia. Nonetheless, the population in New Zealand has declined notably, from average counts of 466 birds during the 1983–1994 period (Sagar et al., 1999) to an average count of 181 birds during the 2005–2019 period (Riegen & Sagar, 2020), a decline of 61%.

Red-necked Stint are arctic migrants that spend the non-breeding season principally in Australia and SE Asia. During the 1983–1994 period Red-necked Stints were the most numerous species in the EAAF with a population of 471,000; this has now been reduced to 315,000, a decline of 33% (Watkins, 1993). In New Zealand the species historically occurred in highest numbers at Lake Ellesmere and in Southland. National counts averaged 158 birds in the 1983–1994 period (Sagar et al., 1999) but decreased to 93 during the 2005–2019 period (Riegen & Sagar, 2020) a decline of 41%.

Sharp-tail Sandpiper is an arctic migrant species with a minimal presence in New Zealand but one that has shown a drastic decline. From an average of 68 birds in the 1983–1994 period (Sagar et al., 1999), numbers declined to an average of 19 birds during the 2005–2019 period (Riegen & Sagar, 2020). This is a decline of 72%, yet the species is not strongly linked to the degradation of the Yellow Sea stopover sites as they are not reliant on intertidal mudflats.

South Island Pied Oystercatcher are a New Zealand breeding shorebird that breeds predominantly inland in parts of the South Island before moving to the coastal habitats during the non-breeding season. Most of the population migrates to the North Island for the winter. South Island Pied Oystercatcher numbers increased through the 20th Century after being protected from hunting in 1940 and by 1970 the national population was estimated at 49,000 birds (Baker 1973). This increase continued to a peak of 112,000 birds in 1994 (Sagar et al., 1999) before a steady decline in numbers to an estimated 79,186 birds in 2019 (Riegen & Sagar, 2020), a decline of 29%.

Pied Stilt are a New Zealand-breeding species predominantly found at coastal and inland wetland habitats, which move to coastal locations over the non-breeding season. The population was estimated at 28,000 birds during the 1983–1994 period based on the winter counts (Sagar et al., 1999) and using the same estimation method the population has shown a 14% decline to 24,000 birds during the 2005–2019 period (Riegen & Sagar, 2020).

Banded Dotterel are an endemic breeding species that breeds in coastal New Zealand and inland including along braided riverbeds before migrating to coasts for the winter. They have a complex migration in which northern breeders tend to move to local coasts, more southern breeders tend to move north within New Zealand, and inland South Island birds tend to migrate to Australia for the winter (Pierce, 1999). During the 1980s and 1990s the population present in New Zealand was thought to be approximately 50,000 birds (Sagar et al., 1999) whereas based on current counts the national population is now thought to be approximately 15,000 birds (Riegen & Sagar, 2020) a 70% decrease since the initial surveys were started. This species is difficult to count accurately as they often flock slightly inland and can easily be missed during counts. The early estimates were based largely on estimates of breeding populations along riverbeds rather than from winter counts, and in fact the estimated New Zealand population from counts in the 1980s and early 1990s was only 10,840 birds (Sagar et al. 1999).

Wrybill are a New Zealand breeding species that breeds exclusively in the South Island before migrating mostly to the northern North Island, especially the greater Auckland region, for the non-breeding season. Wrybill are the only species of those being studied in the project to have shown an increase from the 1983–1994 period with the average count in the 1983–1994 period being 3,657 birds (Sagar et al., 1999) and the average count in the 2005–2019 period 4,769 birds (Riegen & Sagar, 2020), an increase of 30%. This increase can be likely attributed to projects such as Project River Recovery leading to improvement in the quality of braided rivers and the reduction of pests from controlled management schemes, yet at the same time the breeding range may be contracting (Riegen & Dowding, 2003).

My thesis will consist of four chapters, of which this is the first. The second chapter will lay out the methods used in the curating and analysing of the data as well as the structure and timing of the surveys themselves. Chapter 3 will consist of the results from the analyses. Chapter 4 will contain a discussion of the results and what they mean in relation to shorebirds within New Zealand and the greater EAAF, followed by a conclusion summarising the thesis.

My analysis will provide insight into which characteristics play significant roles in the trends of shorebirds within New Zealand such as latitude, site area and intertidal area. The role site characteristics play in the population trends of shorebirds in New Zealand will provide key stakeholders, such as local and national government bodies with a way to evaluate sites that may be 'at risk' allowing a more targeted approach for conservation efforts.

Chapter 2. Methods:

Organizing and curating the data:

Formal shorebird census data were obtained from the Ornithological Society of New Zealand. The counts spanned 1960–2022 (timespans varied between sites) and covered both summer (November–December) and winter (June-July) seasons. Counts were made at 854 local sites nested within 115 main sites, and included counts of 27 species of shorebird, both New Zealand-breeding and international migrants. Counts were amalgamated into the 115 main sites and the frequency and continuity of counts evaluated by plotting all species*site combinations and tabulating count frequencies. Minimum population and count cut-offs were created for each species (i.e., if too few birds or too few counts were present to allow meaningful analysis then the species at that site was not included in the analyses). This reduced the 115 main sites to 17 species of interest and 52 main sites. A final selection process after erroneous count data, such as suspected duplicated counts, were checked led to the final subset to be reduced to 32 sites and 10 species for the analysis carried out.

As the original count data had only positive counts, zero counts were added to all species at each site and season that surveys were undertaken at. The dataset was then checked for errors such as duplicate records and obviously incorrect or implausible counts, such as count numbers for a site displaying a total higher than the entire national population from preceding years. Potential counts errors were checked with the compiler of the data (Adrian Riegen) and corrected as required. The species are listed in Table 1.

Table 1. Species analysed, ordered by migration group (Arctic and New Zealand migrants) and by total population, with seasons and numbers of sites in the final dataset.

Group	Species	Season	Number of sites
Arctic migrants	Bar-tailed Godwit	Summer	32
		Winter	22
	Red Knot	Summer	13
		Winter	4
	Ruddy Turnstone	Summer	13
		Winter	2
	Pacific Golden Plover	Summer	12
	Red-necked Stint	Summer	8
		Winter	2
	Sharp-tailed Sandpiper	Summer	3
New Zealand migrants	South Island Pied Oystercatcher	Winter	17
		Summer	15
	Pied Stilt	Winter	28
		Summer	17
	Banded Dotterel	Winter	23
		Summer	9
	Wrybill	Winter	7
		Summer	2

Basic site details were recorded for all sites in the data, including the longitude and latitude of each site, and the estimated intertidal area. Initially, the values from Hume et al. (2016), Table E1, were used to calculate the intertidal area, but it became obvious that in some cases these estimates were quite erroneous, and they were also available only for true estuaries so did not include all counted sites. Instead, Phil Battley (Zoology and Ecology Group, Massey University) estimated the available intertidal area for each site in the analyses from Google Earth images. This involved making and summing areas from 1–40 polygons of the exposed tidal flats (excluding dense mangrove stands) from recent images taken during low-tide periods. Where no good low-tide images were available (e.g., for most of the Manukau Harbour) approximate tidal flat areas were drawn from bathymetric charts. The value for the

Kaipara Harbour was taken from Swales et al. (2013), who provided estimates of both intertidal area and mangrove cover. The estimate for Lake Ellesmere was checked and updated by Andrew Crossland (Christchurch City Council). Sites are shown in Figure 2; details are given in Table 2.

The two counts, Summer and Winter, differ in the information they gather about the species with the summer counts being an indicator of overall population size for the arctic migrant species as the majority of the population undergoes the migration to New Zealand. The winter counts on the other hand are more of an indicator of productivity with the arctic migrants remaining in New Zealand during the winter season being those of predominantly non-breeding juveniles as they do not migrate back to the breeding grounds in the arctic. This provides a rough estimate as to the breeding success of the past season. This is the opposite for New Zealand's internal migrants which have their breeding season generally inland during the summer season before migrating to larger harbours and estuaries during the winter months, meaning that counts during the winter months on the breeding grounds provide estimates of the population size while the difference between the summer and winter counts provides an indicator of the population's productivity.

Figure 2. Count sites used in species analyses.

Table 2. Locations and estimated intertidal areas accessible to shorebirds for sites included in analyses. Sites are ordered from north to south

Site	Latitude (°S)	Longitude (°E)	Available intertidal area (ha)
Parengarenga Harbour	34.53	172.95	2364
Houhora Harbour	34.78	173.12	770
Rangaunu Harbour	34.95	173.30	4753
Whangarei Harbour	35.80	174.40	1882
Ruakaka	35.90	174.46	44
Waipu	36.00	174.48	80
Mangawhai Estuary	36.09	174.59	246
Whangateau Harbour	36.33	174.77	506
Kaipara Harbour	36.40	174.25	32984
Waitemata Harbour	36.84	174.68	1918
Whitford	36.91	174.96	671
Manukau Harbour	37.05	174.74	19525
Firth of Thames	37.21	175.36	2870
Tauranga Harbour	37.60	176.00	13230
Little Waihi Estuary	37.76	176.48	250
Raglan Harbour	37.80	174.90	1771
Ohiwa Harbour	38.00	177.13	1972
Aotea Harbour	38.01	174.83	2180
Kawhia Harbour	38.08	174.83	4780
Ahuriri Estuary	39.48	176.87	43
Manawatū Estuary	40.47	175.23	76
Farewell Spit	40.52	172.92	6400
Westhaven Inlet	40.60	172.56	1744
Golden Bay	40.67	172.68	3877
Lake Wairarapa	41.23	175.27	161
Avon-Heathcote Estuary	43.55	172.74	585
Lake Ellesmere	43.80	172.63	2097
Otago Harbour	45.79	170.69	2018
Otago Peninsula	45.86	170.68	718
Riverton	46.35	168.01	500
Catlins Lake	46.48	169.70	594
Invercargill	46.49	168.31	1860
Awarua Bay	46.59	168.52	1092
Waikawa	46.63	169.14	531

Choosing the site/species combinations:

Extraction of trend data was a multi-step process. An R-Notebook script was created that generated an html file of all steps and outputs; this was modified for each species/season/site required. First, the data were subset to include just one species/season/site (e.g., Bar-tailed Godwit, Manawatū, summer) and the count data plotted against year in ggplot2, with a LOESS smoother to aid visualisation.

A series of Generalised Additive Models (GAMs) were then run. These use a variable number of curves ("splines") in an attempt to model trends in the data. The selection of k (the number of splines) has to be done manually, and there is no single value of k that can be assumed to be optimal. Selection of a large k-value can lead to over-fitting that represents annual fluctuations rather than long-term trends. The R-script replotted the count data with a GAM (by geom_smooth, method=gam) with k varying from 3–15. Separate GAM models were run for k=3 to k=15 using the gamm function of the mgcv package (Wood, 2017). To evaluate the fit of the models, AIC values for the models were compared to determine if a small k-value had the lowest AIC. If it did not (and in most cases, AIC values became smaller as the number of splines become larger), R^2 values for the models were compared to determine which model explained the variation best.

Once k was selected, the model was checked for appropriateness by the gam.check routine in mgcv. If required (k-index too low) the k-value was changed, and the checks repeated.

Next, the derivSimulCI function was applied to the selected model, and plotted to show periods of significant increase and decrease, where the 95% CI of the trend did not include zero.

Basic statistical summary values were calculated, and trends were extracted for different time periods using the PopTrend package (Knape, 2016). Trends were extracted for standard periods (to the extent of the count data for each site: 1960–2022, 1970–2022, 1980–2022, 1990–2022, 2000–2022 and 2010–2022) and also from the earliest year of counts, the year of maximum modelled count (if there was a population decline; if the population increased the trend would be around zero).

The data were analysed to see whether the median trend was meaningfully different to zero. This was achieved by applying a two-sided non-parametric permutation test (Garren, 2016) to the median trends of each site/season combination for each of the species to generate a p-value that is displayed alongside the histograms of the median trends as seen in Figure 3. This test was not done for species with less than four sites in the analysis.

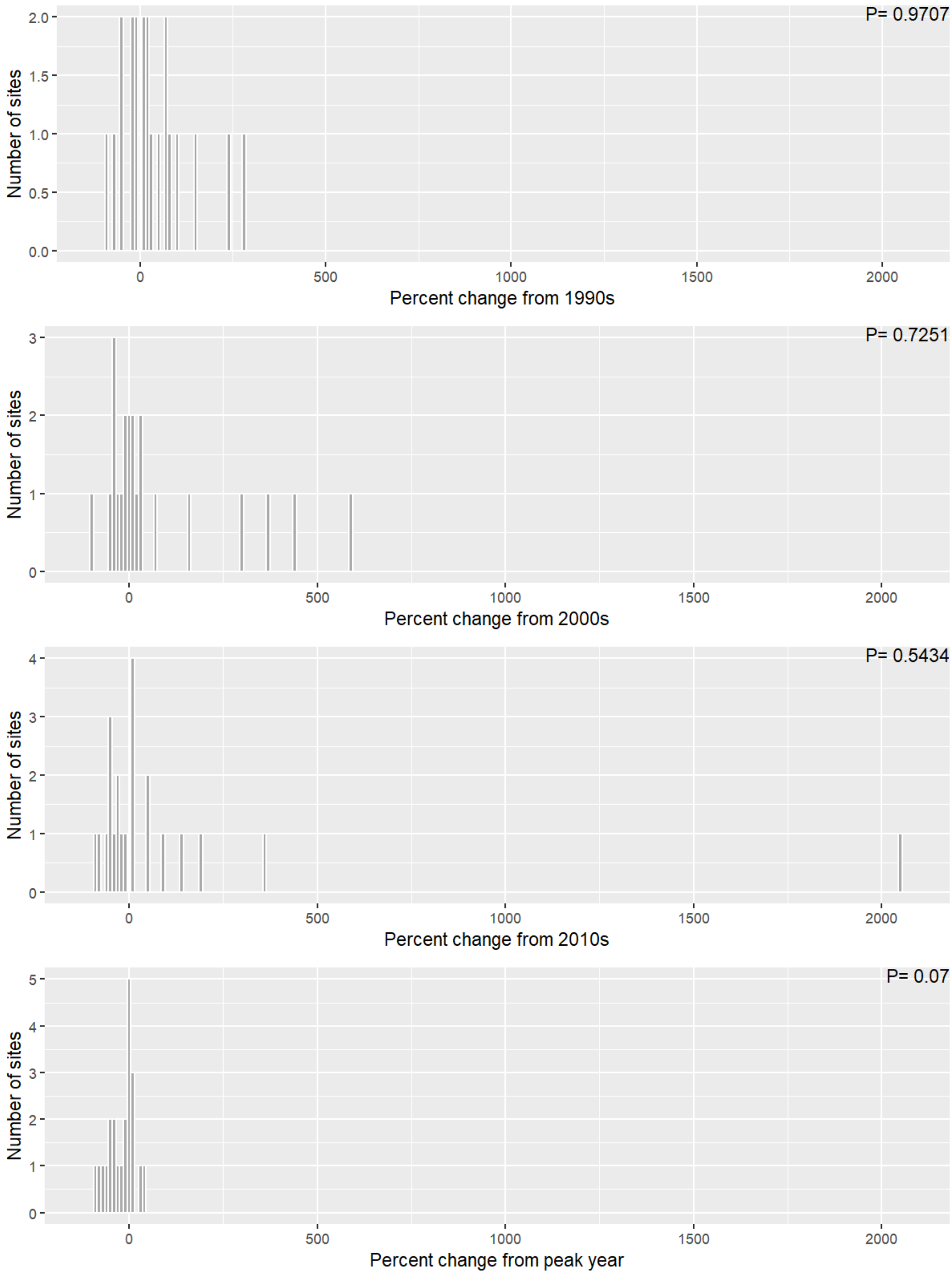


Figure 3. Example of output for the two-sided permutation test used to assess whether the median trend was different to zero.

To test whether a covariate had a significant effect on the trend, linear models were run for each combination of covariates—Trend ~ MaxCount, Trend ~ Latitude, Trend ~ log(Area), Trend ~ MaxCount + Latitude, Trend ~ MaxCount + log(Area), Trend ~ Latitude + log(Area) and lastly Trend ~ MaxCount + Latitude + log(Area). Area was converted to log(Area) to alleviate some of the variation caused by the large difference in scale between the largest and smallest sites. The AIC routine was then used to evaluate which of the generated models had the greatest support, with models with a Delta_AICc value within 2.00 of the best model considered to be meaningful. The plots of the selected model were generated to assess whether there were influential outliers that strongly affected the results. If need be, models were re-run without that value to determine if the results changed in significance. This was repeated for all subsets of the database and to assess all trend variations (Year of Maximum Count, Year of First Count, 2010s, 2000s and 1990).

Note that latitude was coded in the data as degrees south, so a larger value represents a site further south in New Zealand.

Chapter 3. Results:

Arctic Migrants

Bar-tailed Godwits

Summer

Modelled population trends for Bar-tailed Godwits in summer varied from -81% to an increase of 894% depending on the site and time period (Table 3, Figure 4). The median trend was a significant decline for periods of the study from the 1990s, 2000s, and year of maximum count, and almost significantly so from the 2010s (median trends of -15.2% to -38%; significance values are given in Figure 5).

For the three longest surveyed sites (Manukau Harbour, Firth of Thames and Manawatū Estuary) a period of increase spanned the 1970s. This was followed by a decrease in the 1980s and periods of increase and decrease in the 1990s and 2000s, respectively, for both Manukau Harbour and Firth of Thames. Manawatū Estuary differed from the other two longstanding surveyed sites as it showed a near constant increase from the 1960s until the end of the 1990s before entering a constant decline that continues to the present day. Other sites covering shorter time periods show a downturn in population around 2005 and the start of the 2010s.

The magnitude of change at sites with large populations indicates there have been major population size changes over time. This has involved virtually halving of the location populations at sites including the Kaipara Harbour (from ~20,000 to 10,000 birds), Manukau Harbour (cycles from ~20,000 to 10,000), Firth of Thames (~10,000 to 5,000), Tauranga Harbour (~8,000 to 4,000), Ohiwa Harbour (4,000 to 2,000), Kawhia Harbour (4,000 to 2,000) and the Avon-Heathcote Estuary (2,000 to 1,000).

This is reinforced by the results of linear models testing whether trends at individual sites varied with population size, habitat area and latitude. For all trends, the best model (lowest AIC model; Table 4) found that the population trend varied positively with latitude. Sites at higher latitudes (further south) had higher trends (i.e., greater increases); this trend was significant or almost so for all analysed time periods.

Table 3. Bar-tailed Godwit summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information.

Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Houhora Harbour	13	5	1300	711	724.77	-41 (-69, 15)	-39 (-65, 3)	-30 (-56, 8.7)	-30 (-56, 8.7)	-41 (-68, 14)
Rangaunu Harbour	11	7	4000	3500	2987.6	-60 (-77, -34)	-71 (-92, -16)	-51 (-73, -20)	-63 (-78, -40)	-61 (-77, -36)
Whangarei Harbour	26	10	5301	2863	2704.6	-38 (-77, 74)	-66 (-85, -26)	-54 (-79, 4.7)	-62 (-83, -17)	-40 (-77, 63)
Waipu	24	12	163	102	102.9	-42 (-64, -2.5)	-18 (-45, 25)	-6.6 (-42, 47)	-14 (-44, 33)	-41 (-63, -2.6)
Mangawhai Estuary	24	7	600	323.5	322	-0.32 (-37, 62)	-0.25 (-30, 45)	-0.13 (-17, 22)	-0.27 (-33, 51)	-0.32 (-37, 62)
Kaipara Harbour	27	10	21823	14010	14322.6	-46 (-64, -19)	-46 (-63, -19)	-25 (-49, 8.6)	-44 (-62, -17)	-45 (-63, -18)
Waitemata Harbour	18	6	3487	247	716.3	-80 (-97, 44)	73 (-96, 84)	138 (-82, 2185)	-78 (-96, 53)	-79 (-97, 44)
Whitford	14	9	4868	835	1154.4		66 (-65, 782)	53 (-51, 386)	5.6 (-12, 25)	66 (-65, 759)
Manukau Harbour	62	11	24865	14163	14728.4	-40 (-57, -16)	-38 (-56, -12)	-36 (-57, -5.8)	-47 (-63, -25)	-32 (-55, 3.2)
Firth of Thames	62	11	14620	7071	7446.3	24 (-21, 96)	8.3 (-33, 71)	16 (-30, 87)	-22 (-51, 18)	-16 (-52, 46)
Tauranga Harbour	22	8	10200	5740.5	5914.7	14 (-44, 127)	-16 (-51, 39)	-34 (-58, 4.8)	-30 (-54, 4.5)	12 (-44, 121)
Little Waihi Estuary	19	8	1350	911	856.05	-39 (-61, 5.3)	-40 (-59, -8.1)	-33 (-52, -4.1)	-39 (-59, -9.9)	-39 (-61, -5.6)
Raglan Harbour	16	11	1055	550	521.69		-14 (-65, 112)	-9.4 (-51, 65)	-10 (-54, 73)	-14 (-65, 111)
Ohiwa Harbour	26	14	5000	2846.5	3060.3	-57 (-66, -44)	-52 (-62, -38)	-35 (-48, -19)	-54 (-64, -40)	-59 (-68, -46)
Aotea Harbour	24	12	2950	1440.5	1423.1	-53 (-81, 3.8)	-59 (-81, -17)	-52 (-77, -4.1)	-59 (-81, -17)	-54 (-81, 1.5)
Kawhia Harbour	25	10	6678	2647	2875.8	-49 (-70, -20)	-40 (-60, -16)	-23 (-37, -8.3)	-42 (-61, -16)	-49 (-69, -20)
Ahuriri Estuary	25	7	312	178	177.1	-21 (-53, 36)	-17 (-44, 27)	-9.1 (-26, 13)	-20 (-52, 34)	-21 (-53, 36)
Manawatū Estuary	60	10	570	400	383.6	-61 (-71, -50)	-54 (-65, -39)	-27 (-44, -2.9)	-58 (-69, -46)	-20 (-43, 13)
Farewell Spit	28	14	22923	9556	10262.5	-37 (-59, -1.9)	-12 (-40, 32)	-2.7 (-35, 43)	-32 (-54, 3.6)	-36 (-57, -2.7)
Westhaven Inlet	21	7	1850	830	977.8		-54 (-75, -25)	-55 (-72, -26)	-53 (-72, -22)	-54 (-74, -26)
Golden Bay	26	12	2285	1097	1098	22 (-31, 134)	17 (-25, 93)	8.5 (-14, 41)	17 (-25, 93)	22 (-31, 132)
Lake Wairarapa	12	4	117	55.5	52.3		7.4 (-69, 257)	-21 (-65, 85)	-24 (-66, 73)	6.3 (-69, 248)
Avon-Heathcote Estuary	23	10	2088	1441	1470	9.2 (-18, 46)	-7.5 (-26, 15)	-35 (-47, -20)	-35 (-47, -20)	8.1 (-17, 43)
Lake Ellesmere	24	9	443	127.5	142.7	417 (85, 1469)	254 (61, 733)	94 (28, 203)	83 (25, 174)	408 (84, 1427)
Otago Harbour	18	8	720	195.5	201.3	-71 (-96, 103)	-81 (-98, 74)	43 (-87, 1084)	-80 (-98, 34)	-71 (-96, 91)
Otago Peninsula	18	6	1058	629	593.6	20 (-64, 303)	-26 (-70, 91)	-25 (-70, 70)	-40 (-75, 42)	17 (-64, 291)
Riverton	11	8	507	360	320		87 (-27, 344)	290 (37, 899)	14 (-46, 160)	117 (-9.9, 384)
Catlins Lake	17	6	443	302	302.7	-3.9 (-38, 51)	14 (-20, 61)	29 (-3.2, 67)	0 (0, 0)	-3.2 (-37, 51)
Invercargill	18	8	2570	817.5	1120	194 (14, 765)	894 (309, 2652)	72 (-7.3, 229)	0 (0, 0)	219 (29, 798)
Awarua Bay	19	8	1142	712	729	-5.6 (-53, 78)	-0.61 (-43, 73)	7 (-27, 55)	-5.3 (-53, 78)	-5.3 (-53, 78)

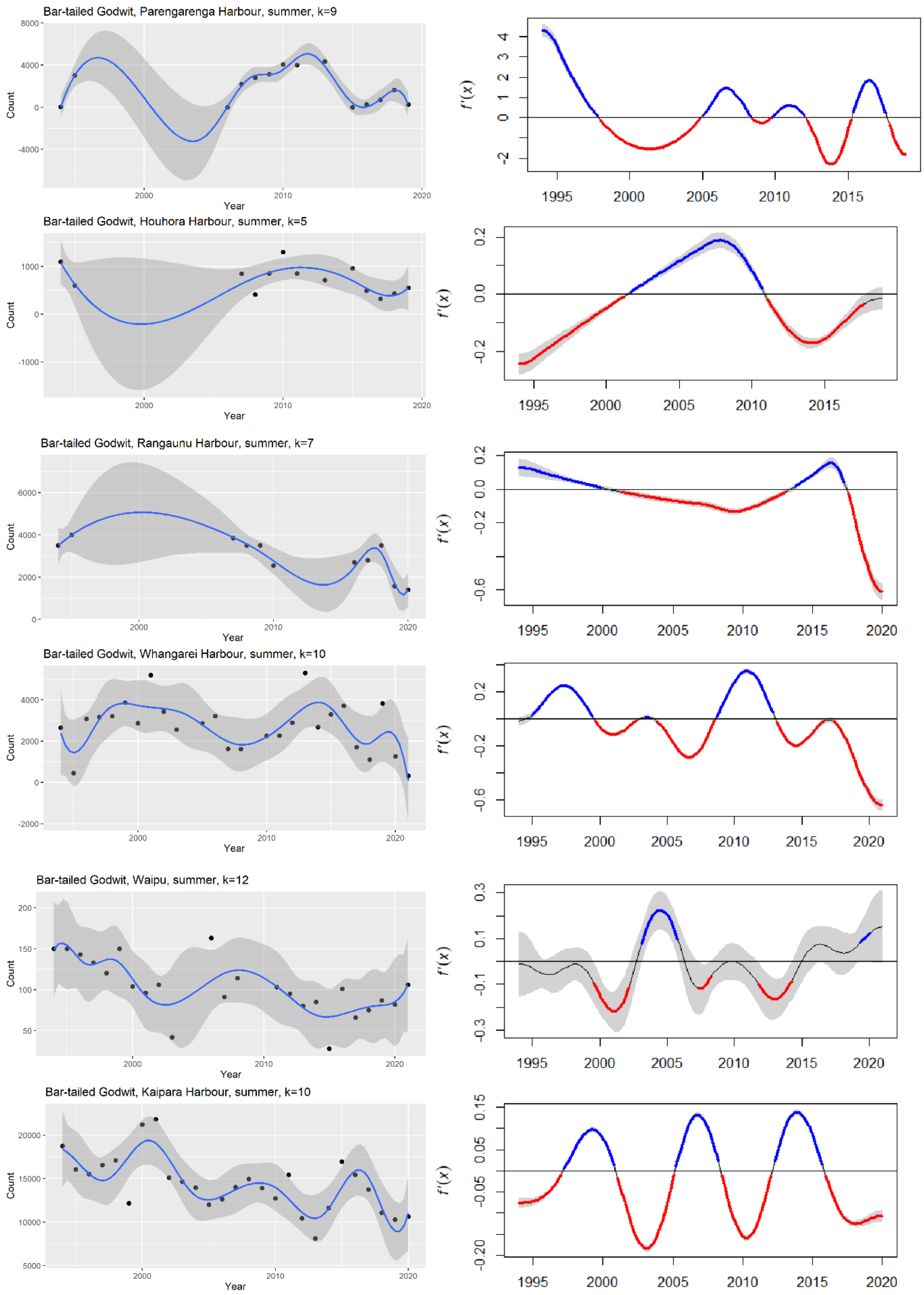
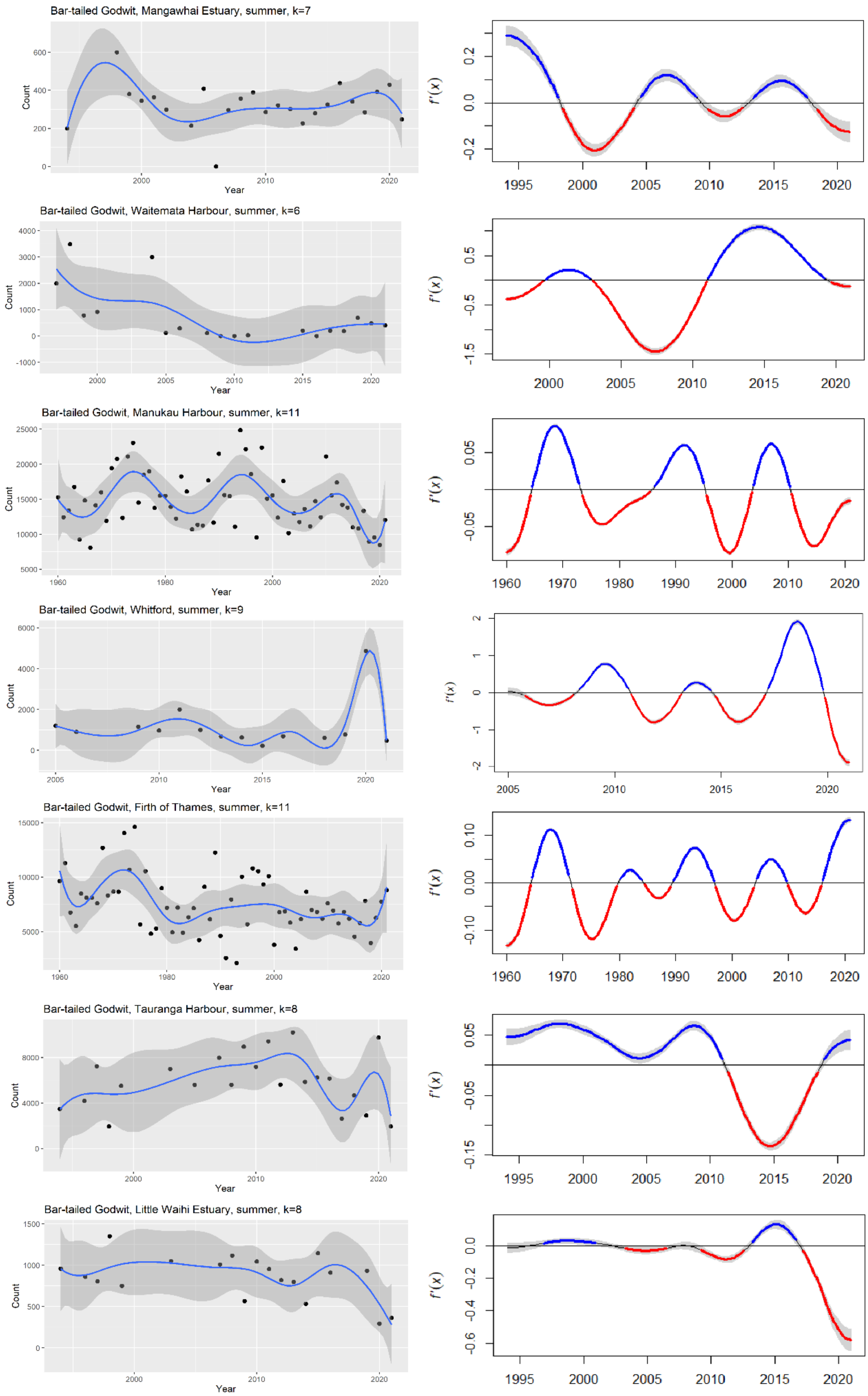


Figure 4. Site-specific trends of Bar-tailed Godwits in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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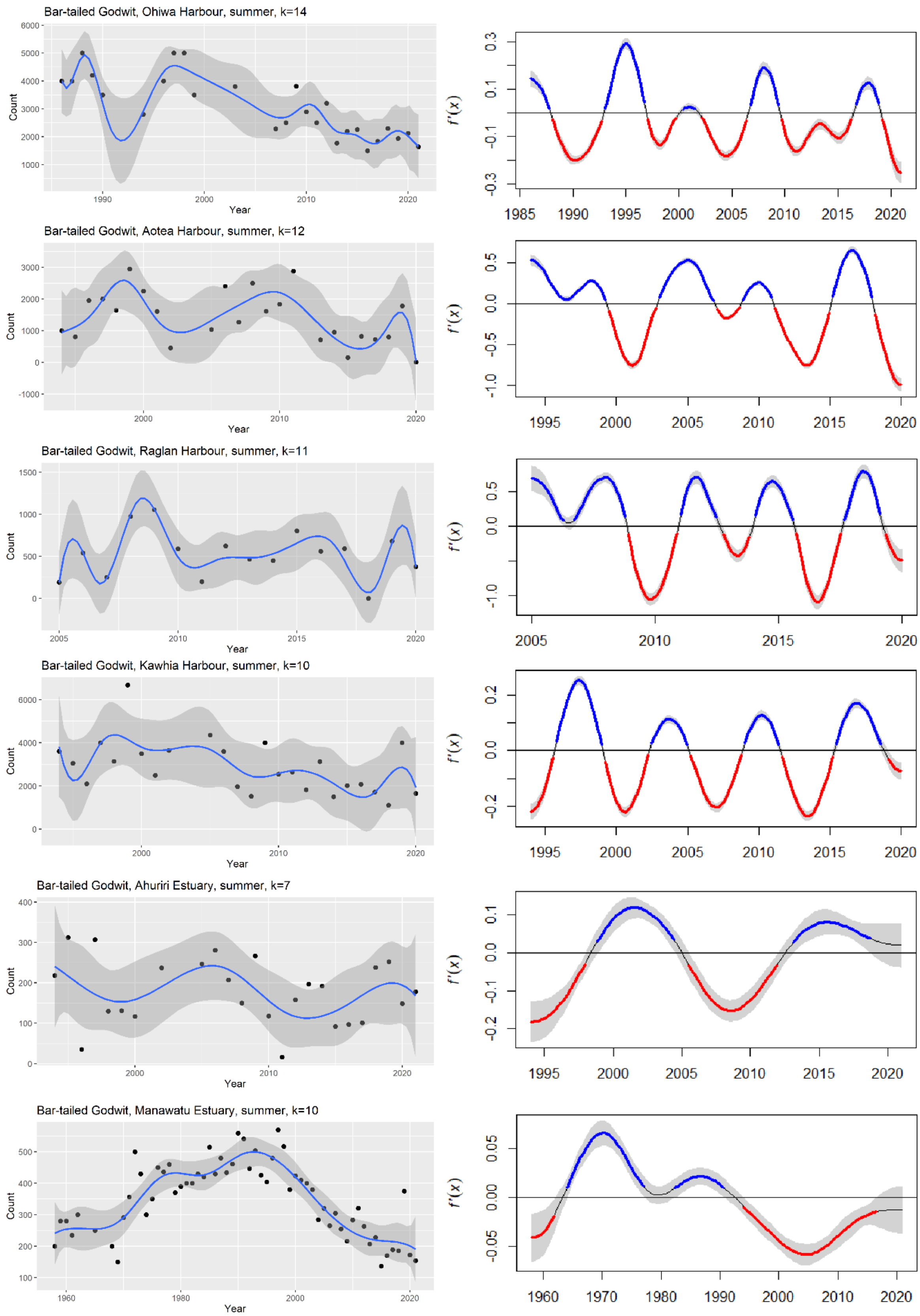

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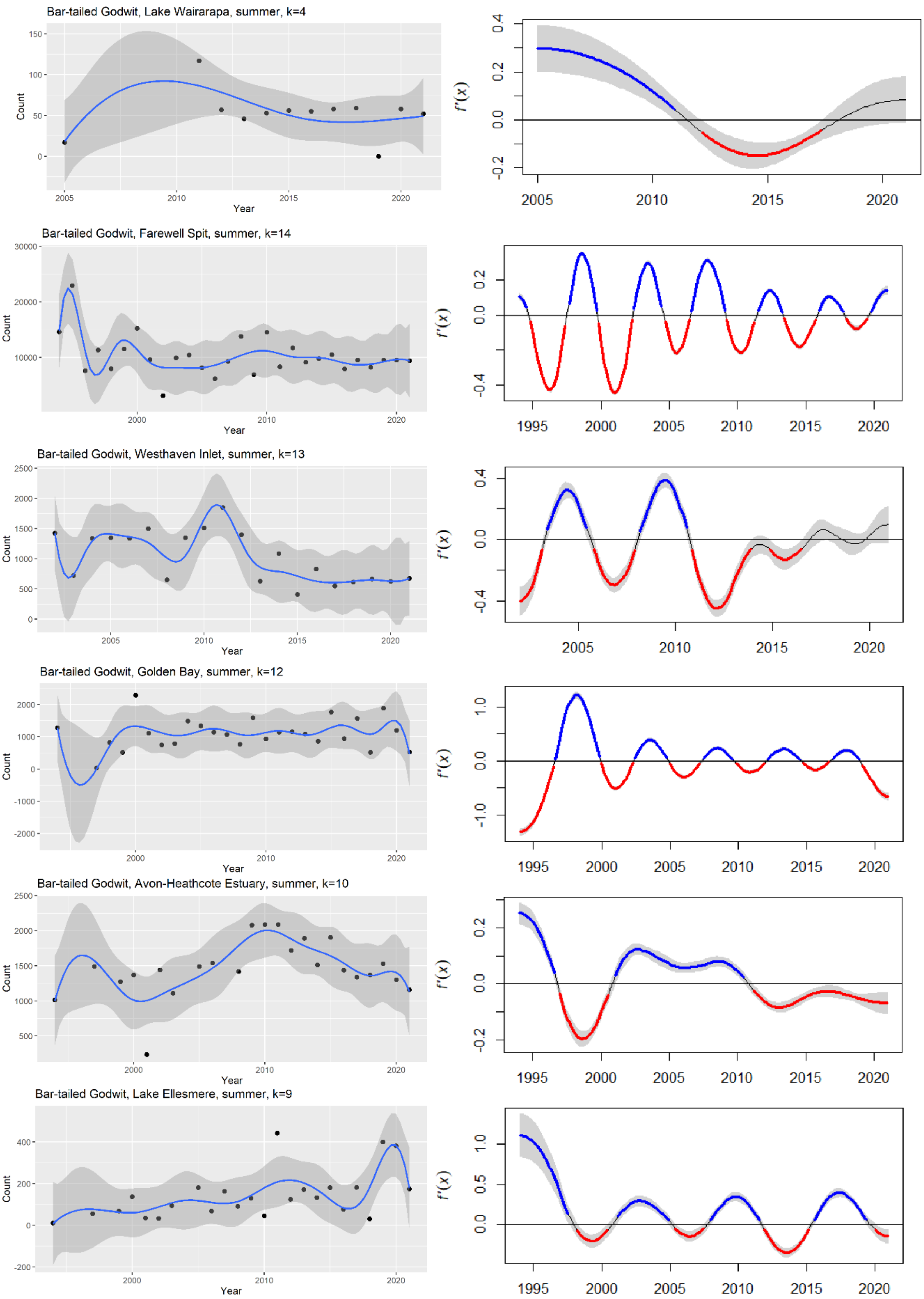

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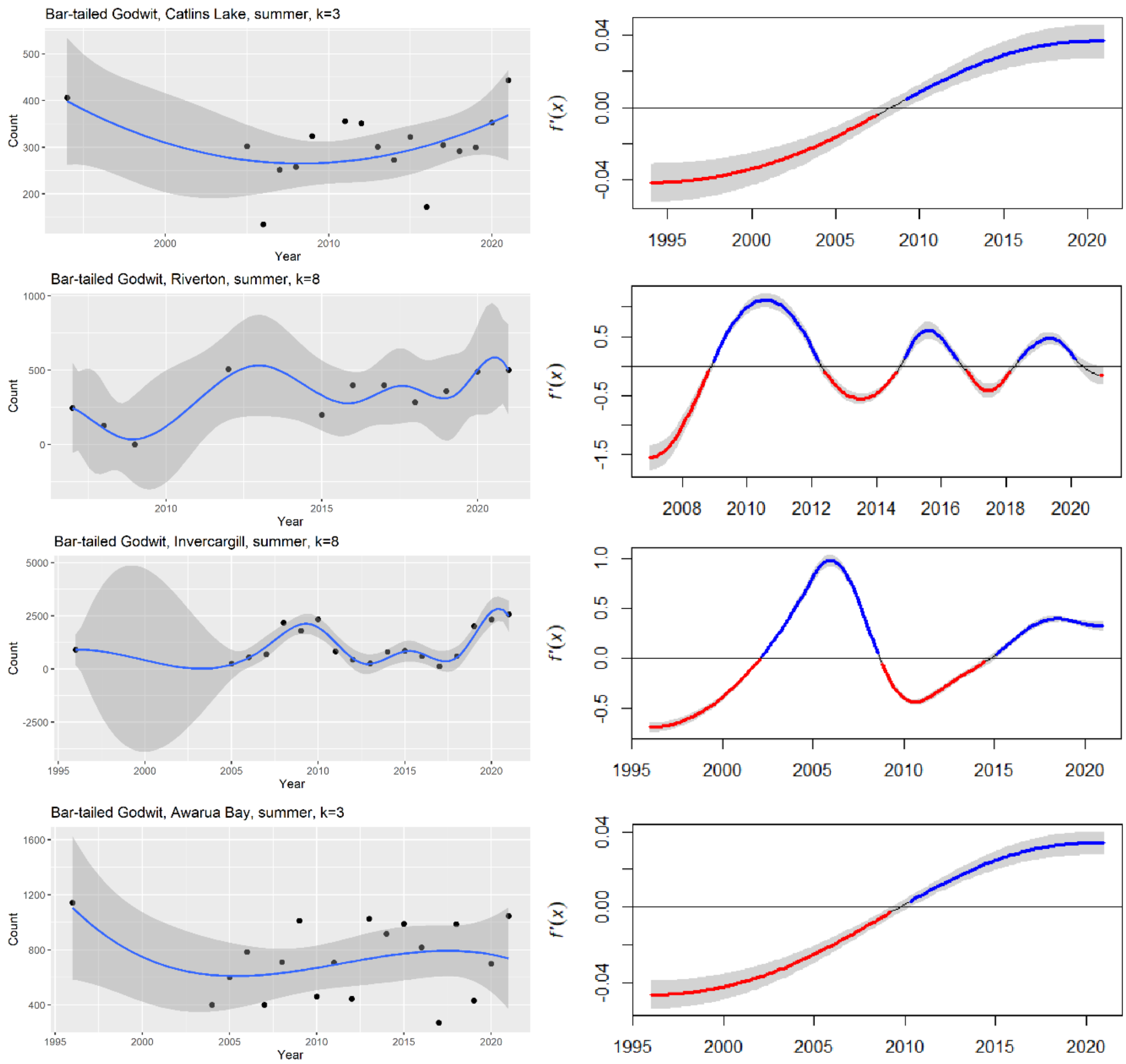


Figure 4 Continued

Table 4. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Bar-tailed Godwits in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.53	Intercept	-431.611	193.154	-2.235	0.0355
	$F_{1,23}$=5, P=0.03533, R^2=0.1429		Latitude	10.808	4.833	2.236	0.0353
2000s	Lat	0.55	Intercept	-695.38	313.92	-2.215	0.0351
	$F_{1,28}$=5.275, P=0.02932, R^2=0.1285		Latitude	17.98	7.83	2.297	0.0293
2010s	Lat	0.57	Intercept	-303.916	122.740	-2.476	0.0196
	$F_{1,28}$=6.444, P=0.01698, R^2=0.158		Latitude	7.772	3.062	2.538	0.0170
MaxYr	Lat	0.42	Intercept	-140.185	61.340	-2.285	0.0301
	$F_{1,28}$=3.41, P=0.07531, R^2=0.07679		Latitude	2.826	1.530	1.847	0.0753

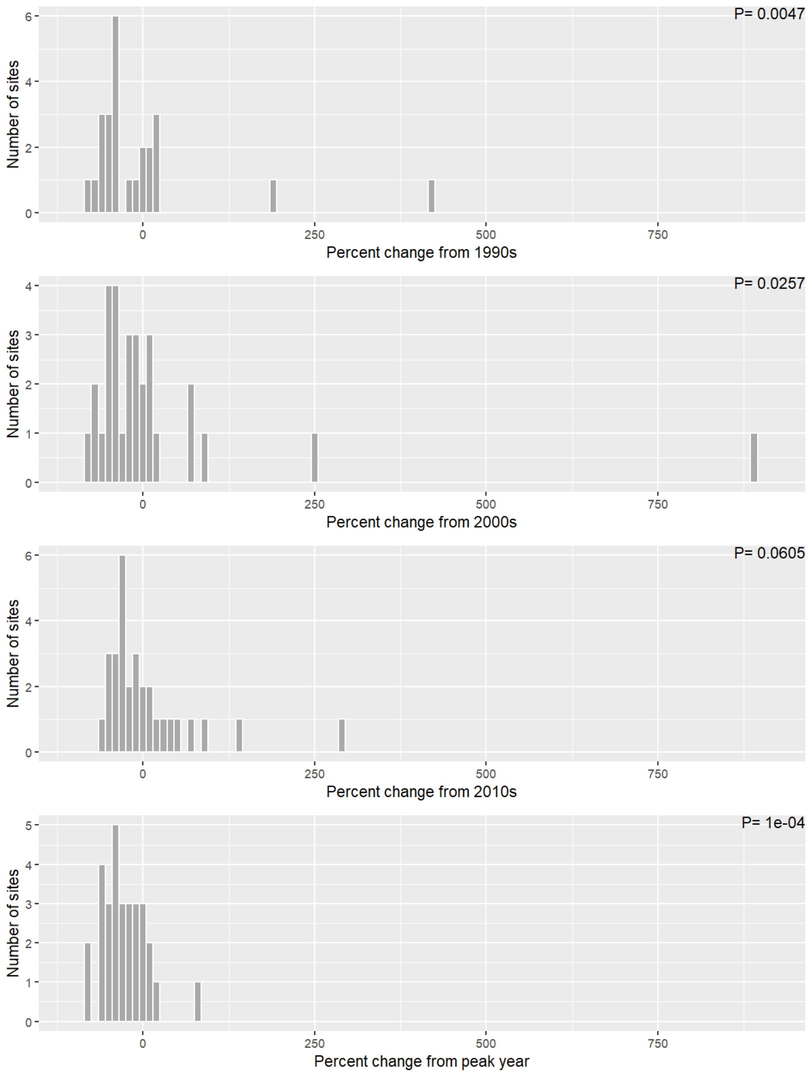


Figure 5. Individual site trends for Bar-tailed Godwits in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Winter

Modelled population trends for Bar-tailed Godwits in winter varied from -99% to an increase of 2052% depending on the site and time period (Table 5, Figure 6). The median trend was not different to zero for the periods from the 1990s, 2000s or 2010s (median trends of 0.4% to 19%; significance values are given in Figure 7), but it was indicatively lower than zero for the trends from the year of maximum count (-7.15%, P=0.07).

Sites that were counted from the 1990s showed a drastic decline from a peak in 1990 with both Manukau Harbour and Ohiwa Harbour declining to half of their peak count numbers that never return to the peak numbers (Figure 6). Other sites showed a common trend of the years immediately preceding 2010 having a decline for North Island sites, while most southern sites appear to show this period as an increase. Most sites showed an increasing trend from 2019 onwards.

This is reinforced by the results of linear models testing whether trends at individual sites varied with population size, area and latitude. For all trends except for the year of the maximum count, the best model (lowest AIC model; Table 6) included latitude as a factor, though it was only marginally significant ($p < 0.1$).

Table 5. Bar-tailed Godwit winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Parengarenga Harbour	9	6	796	349	350.7	-91 (-99, 7.7)	-95 (-99, -21)	-90 (-98, -4.9)	-94 (-99, -24)	-92 (-99, 2)
Rangaunu Harbour	9	6	593	194	224.2	-8.9 (-93, 953)	-36 (-94, 421)	-46 (-89, 174)	-41 (-84, 141)	-10 (-93, 922)
Whangarei Harbour	26	14	472	274	281.7	12 (-56, 176)	1 (-51, 95)	95 (-5.9, 299)	13 (-45, 140)	11 (-54, 163)
Waipu	24	8	14	0.5	3.3	279 (-60, 3291)	302 (-19, 1845)	364 (-6.9, 2883)	37 (-72, 552)	288 (-54, 3115)
Kaipara Harbour	27	10	3247	1555	1516.7	239 (41, 813)	67 (-13, 217)	7.9 (-42, 95)	-7.4 (-46, 68)	195 (33, 637)
Manukau Harbour	61	13	5992	1926	2213.2	-65 (-83, -25)	-26 (-64, 55)	-38 (-72, 26)	-65 (-83, -25)	-42 (-76, 34)
Firth of Thames	28	14	1372	596	669.3	68 (-36, 275)	19 (-33, 129)	52 (-16, 178)	31 (-26, 164)	65 (-33, 253)
Tauranga Harbour	20	9	1908	683.5	795.9	50 (-33, 255)	4.5 (-44, 103)	-32 (-65, 30)	-32 (-61, 20)	47 (-33, 246)
Little Waihi Estuary	19	10	169	34	47.8	-16 (-77, 223)	-13 (-69, 156)	-6.9 (-46, 65)	-6.9 (-46, 65)	-15 (-76, 1`8)
Ohiwa Harbour	24	14	700	335	365	-49 (-77, 1.7)	-19 (-64, 69)	-19 (-66, 66)	-43 (-72, 18)	-27 (-68, 57)
Aotea Harbour	24	9	231	95.5	95.4	-48 (-78, 39)	-50 (-77, 13)	-47 (-74, 17)	-51 (-77, 15)	-48 (-78, 38)
Kawhia Harbour	25	13	806	212	280.7	-8.5 (-77, 308)	-35 (-79, 114)	-63 (-87, 21)	-63 (-87, 21)	-10 (-77, 279)
Ahuriri Estuary	26	8	71	25	26.5	6.3 (-56, 150)	9.2 (-46, 122)	11 (-32, 69)	1.4 (-4.1, 7.1)	6.4 (-55, 148)
Farewell Spit	26	11	2626	1703.5	1698.7	82 (13, 193)	33 (-8.5, 92)	52 (1.9, 119)	0 (0, 0)	75 (15, 169)
Westhaven Inlet	19	14	300	46	69		589 (-54, 9116)	-78 (-95, 9)	-79 (-96, -3.6)	507 (-57, 7269)
Golden Bay	26	15	352	80.5	95.9	19 (-65, 295)	15 (-57, 200)	7.7 (-36, 78)	14 (-56, 185)	19 (-65, 290)

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Avon-Heathcote Estuary	23	5	410	203	210.52	-16 (-54, 47)	-43 (-65, -5.3)	-50 (-68, -18)	-50 (-68, -18)	-19 (-54, 40)
Otago Harbour	18	11	193	28.5	53.8	23 (-85, 660)	-7 (-81, 317)	-26 (-76, 117)	-23 (-69, 85)	21 (-85, 609)
Otago Peninsula	18	10	229	89	92.3	34 (-57, 330)	26 (-49, 221)	13 (-30, 85)	8.1 (-20, 48)	33 (-57, 324)
Catlins Lake	14	5	68	23	21.6	100 (-55, 827)	160 (-20, 797)	193 (14, 679)	0 (0, 0)	103 (-54, 825)
Invercargill	17	10	375	83	111.7	73 (-18, 291)	440 (-11, 4846)	2052 (579, 6830)	0 (0, 0)	85 (-5.4, 283)
Awarua Bay	19	11	244	63	76.1	153 (-42, 942)	366 (-13, 2404)	139 (-10, 516)	0 (0, 0)	163 (-35, 898)

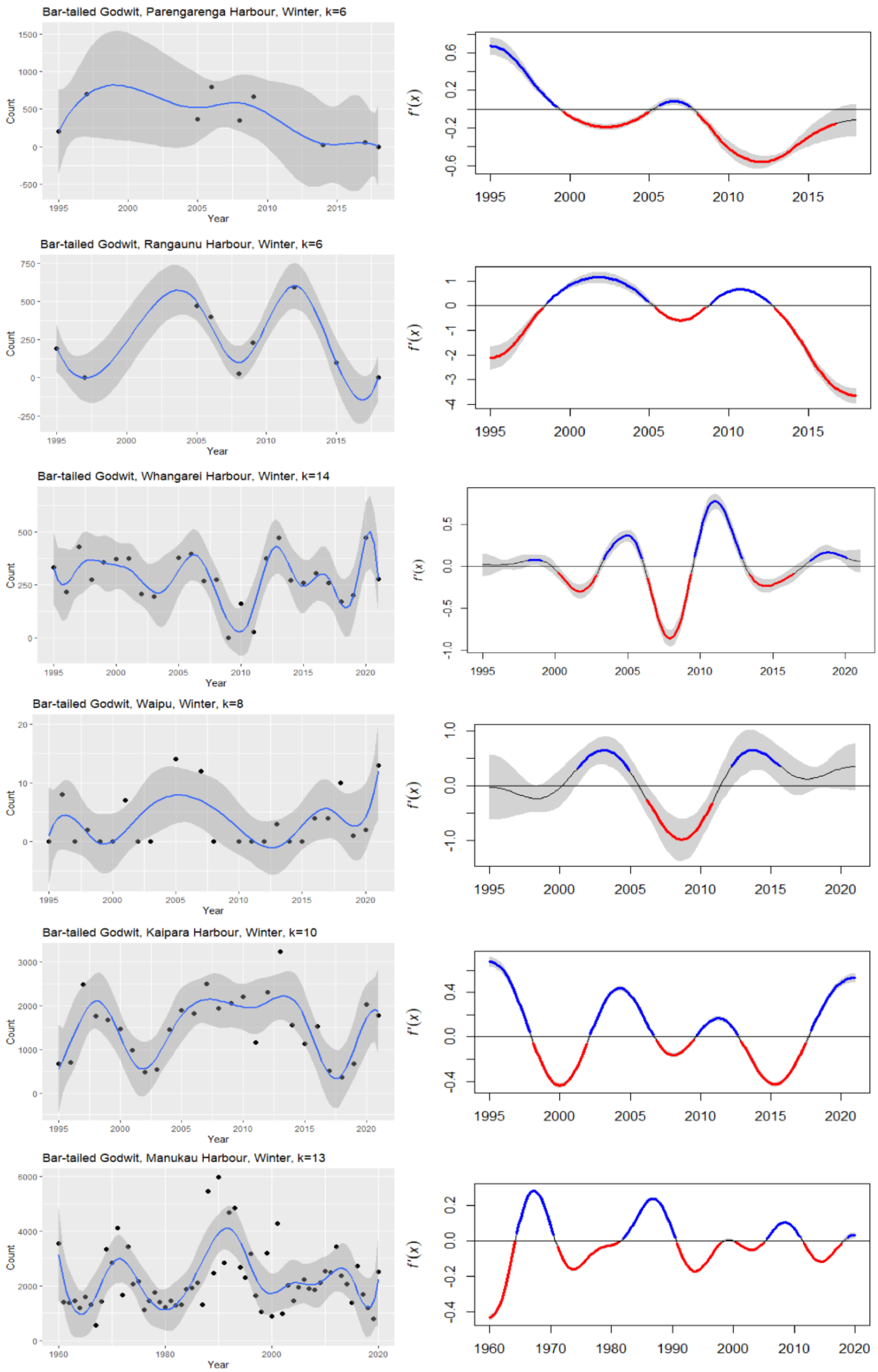
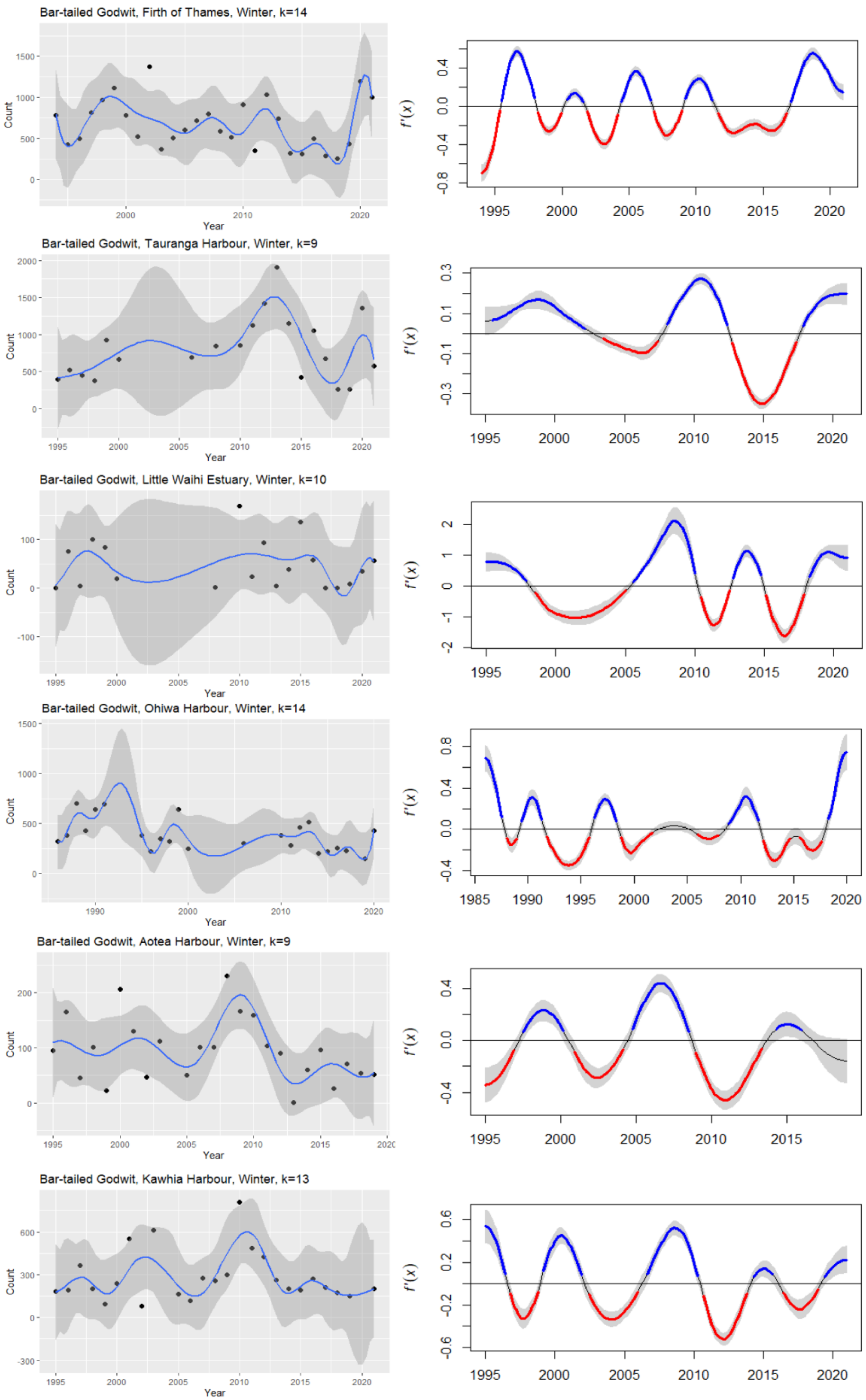


Figure 6. Site-specific trends of Bar-tailed Godwits in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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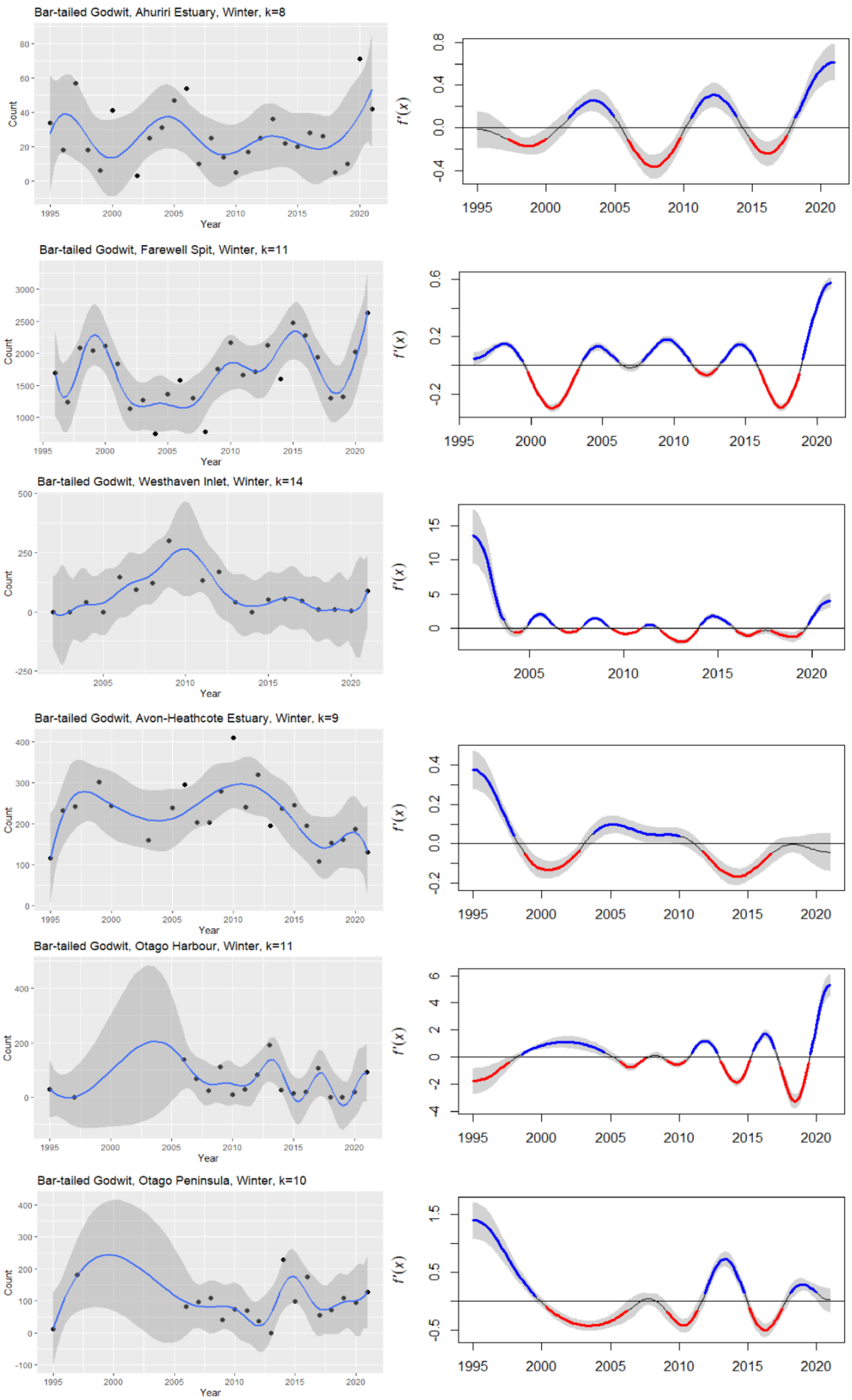

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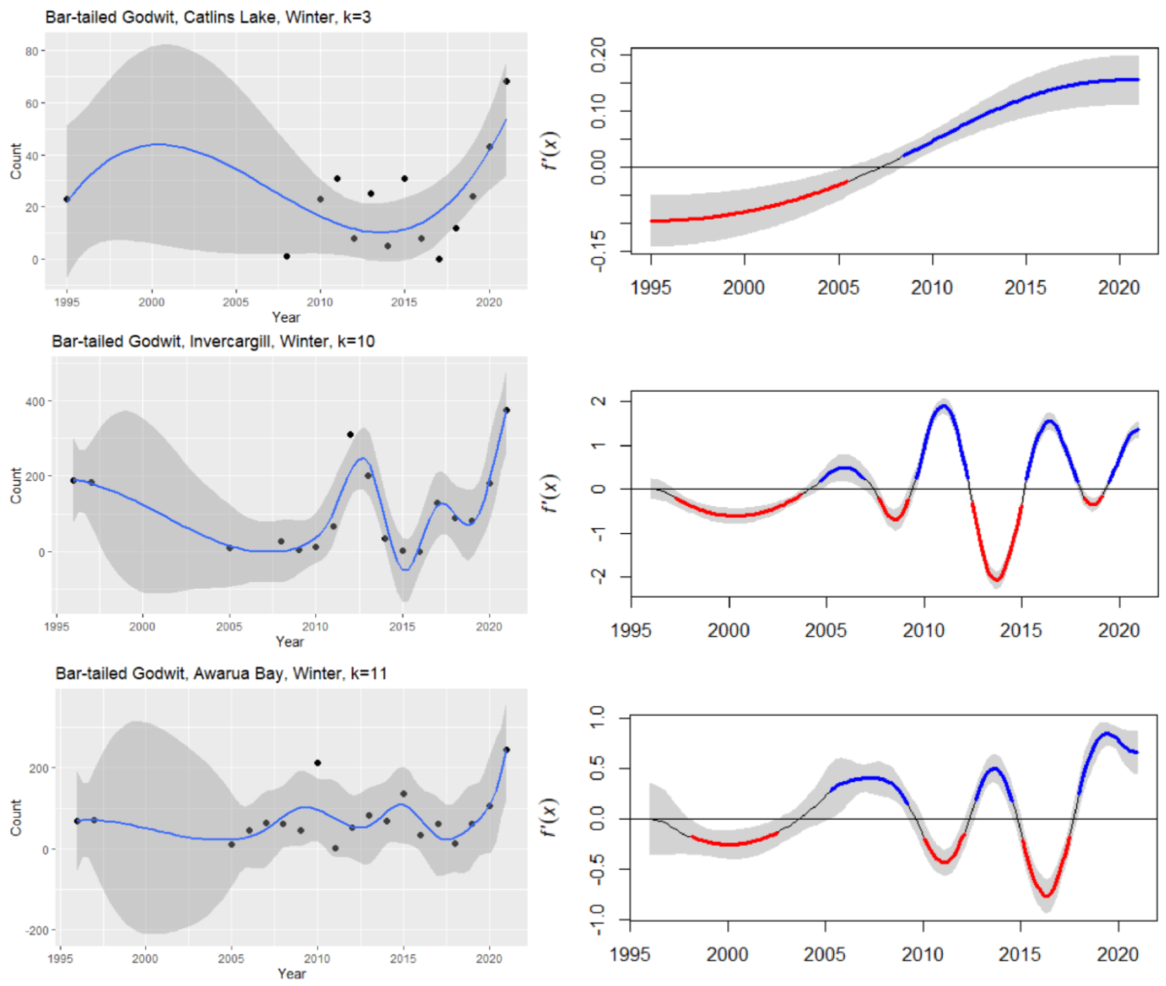


Figure 6 Continued

Table 6. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Bar-tailed Godwits in winter. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.30	Intercept	-100.222	202.729	-0.494	0.627
	$F_{1,19}$=0.4821, P=0.4959, R^2=-0.02658		Latitude	3.514	5.061	0.694	0.496
2000s	Lat	0.49	Intercept	-644.21	361.12	-1.784	0.0896
	$F_{1,20}$=4.036, P=0.05824, R^2=0.1263		Latitude	18.10	9.01	2.009	0.0582
2010s	Lat	0.52	Intercept	-1557.02	902.31	-1.726	0.0998
	$F_{1,20}$=3.461, P=0.07761, R^2=0.1049		Latitude	41.88	22.51	1.860	0.0776
MaxYr	Log(Area.ha)	0.39	Intercept	37.270	36.112	1.032	0.314
	$F_{1,20}$=2.669, P=0.1179, R^2=0.07364		Log(Area.ha)	-7.707	4.717	-1.634	0.118

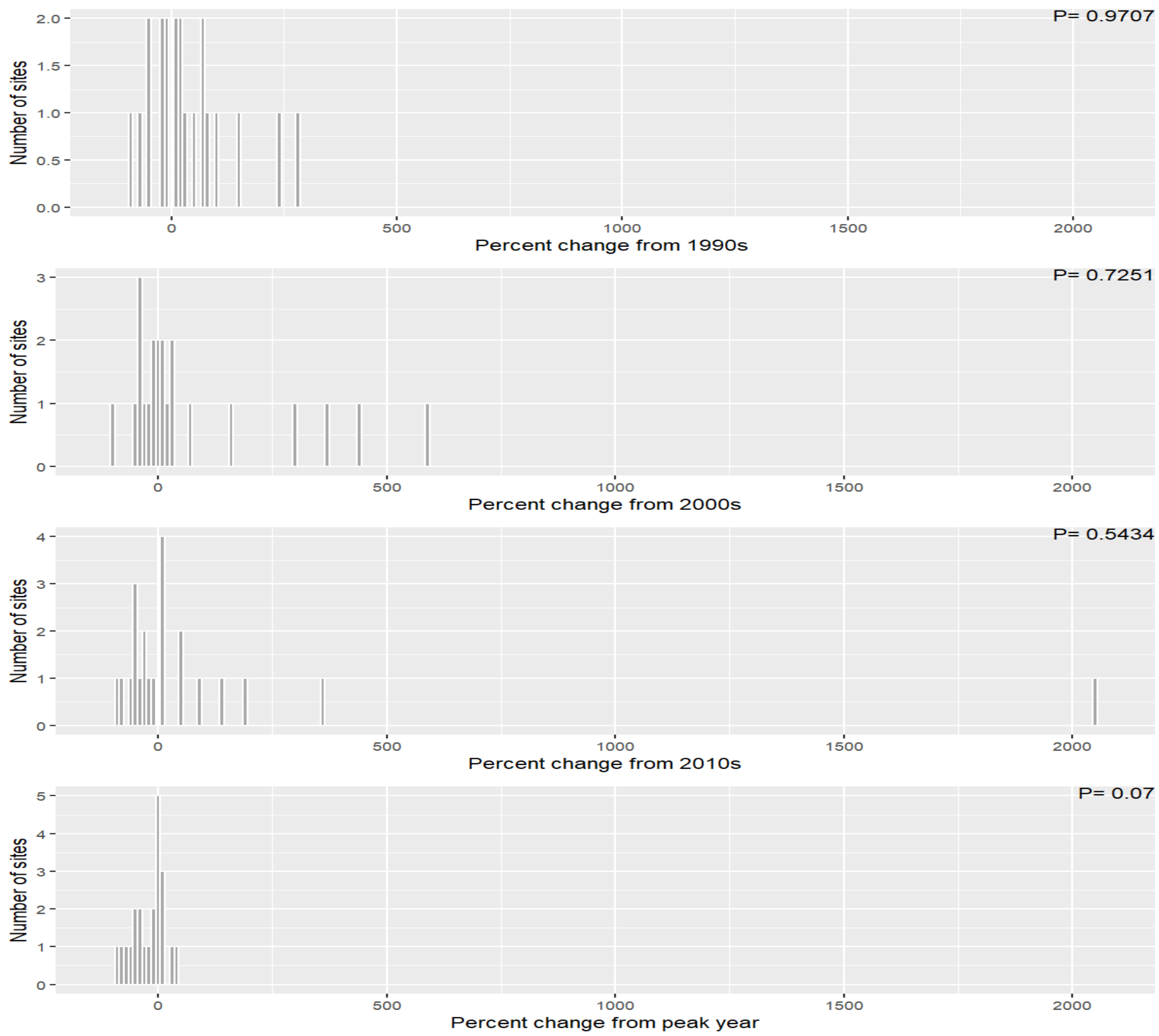


Figure 7. Individual site trends for Bar-tailed Godwits in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Red Knot

Summer

Modelled population trends for Red Knot in Summer varied from -99% to an increase of 590% depending on the site and time period (Table 7, Figure 8). The median trend was significantly lower than zero for all periods of the study except from the 1990s, when a single site had a large increase from a small initial count (Figure 9). Median trends of ranged from -34% to -71.

Only three sites have not declined since the first year of counts, and even those sites (Mangawhai Estuary, Manukau Harbour and Manawatū Estuary) have experienced periods of substantial increase and decrease. The overall population has likely declined, as three of the largest population sites (Kaipara Harbour, Firth of Thames and Farewell Spit) have decreased substantially. There were no significant relationships between trends at individual sites and population size, habitat area or latitude (Table 8).

Table 7. Red Knot summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	TrendYr1
Houhora Harbour	13	9	1600	230	366.5	-92 (-98, -72)	-83 (-95, -43)	-64 (-90, 23)	-92 (-98, -71)	-92 (-98, -71)
Rangaunu Harbour	12	7	7000	582	1391.2	-88 (-98, -55)	-86 (-98, -3.8)	129 (-59, 1143)	-88 (-97, -54)	-88 (-97, -56)
Whangarei Harbour	26	12	3050	1125	1229.8	-79 (-90, -54)	-70 (-83, -45)	-47 (-61, -27)	-73 (-86, -48)	-79 (-90, -54)
Mangawhai Estuary	24	13	380	195.5	193	590 (5.4, 3933)	3.8 (-47, 104)	68 (-18, 243)	-7.4 (-35, 38)	498 (1.1, 3038)
Kaipara Harbour	27	14	14146	8398	8262.8	-53 (-76, -7.2)	-40 (-68, 7.2)	-1.8 (-48, 79)	-50 (-73, -6)	-53 (-76, -11)
Whitford	14	10	1000	596.5	527.1		-41 (-80, 96)	-31 (-66, 58)	-41 (-79, 95)	-41 (-79, 95)
Manukau Harbour	62	14	23200	9400.5	9739.6	-56 (-75, -27)	-18 (-51, 42)	-20 (-52, 38)	-54 (-74, -23)	253 (13, 1056)
Firth of Thames	62	15	11600	4265	4884.5	-46 (-72, 6.4)	-62 (-81, -24)	-44 (-73, 11)	-71 (-85, -44)	-67 (-85, -29)
Little Waihi Estuary	19	9	104	32	35.9	-99 (-100, -61)	-97 (-100, -24)	-63 (-100, 1958)	-99 (-100, -67)	-99 (-100, -60)
Manawatū Estuary	61	15	430	199	172.9	-89 (-94, -81)	-86 (-92, -75)	-76 (-87, -56)	-90 (-94, -82)	99 (-22, 437)
Farewell Spit	28	12	15591	8162	8262.4	-37 (-61, 0.88)	16 (-23, 73)	-0.73 (-37, 55)	-35 (-59, 2.9)	-35 (-59, 2.9)
Westhaven Inlet	21	9	215	37	65		-77 (-95, 14)	-55 (-81, 7.7)	-75 (-94, 13)	-77 (-95, 14)
Lake Ellesmere	24	14	57	9	17.2	-65 (-97, 344)	32 (-85, 1127)	-34 (-92, 514)	-64 (-95, 160)	-59 (-96, 356)

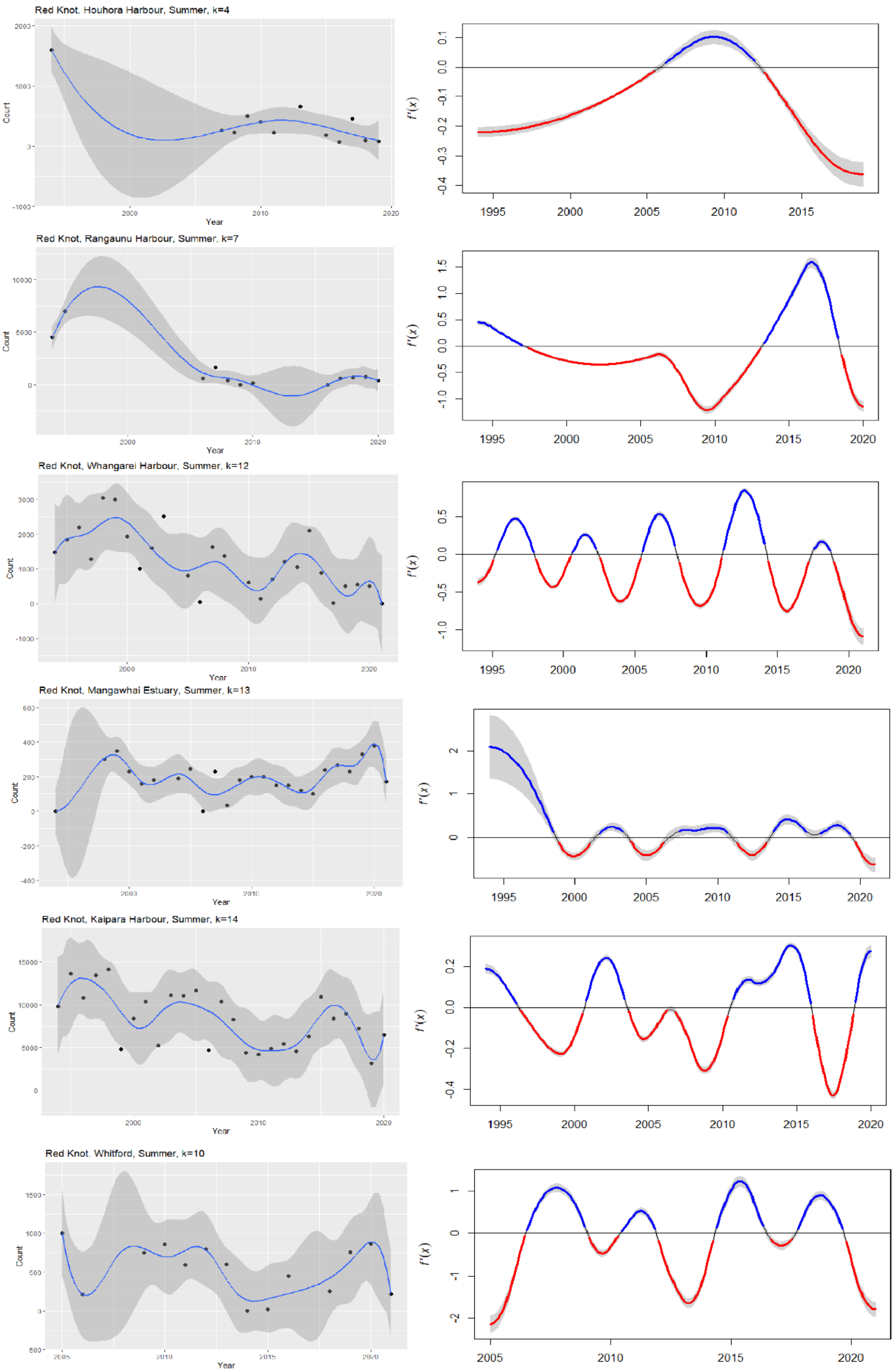


Figure 8. Site-specific trends of Red Knots in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

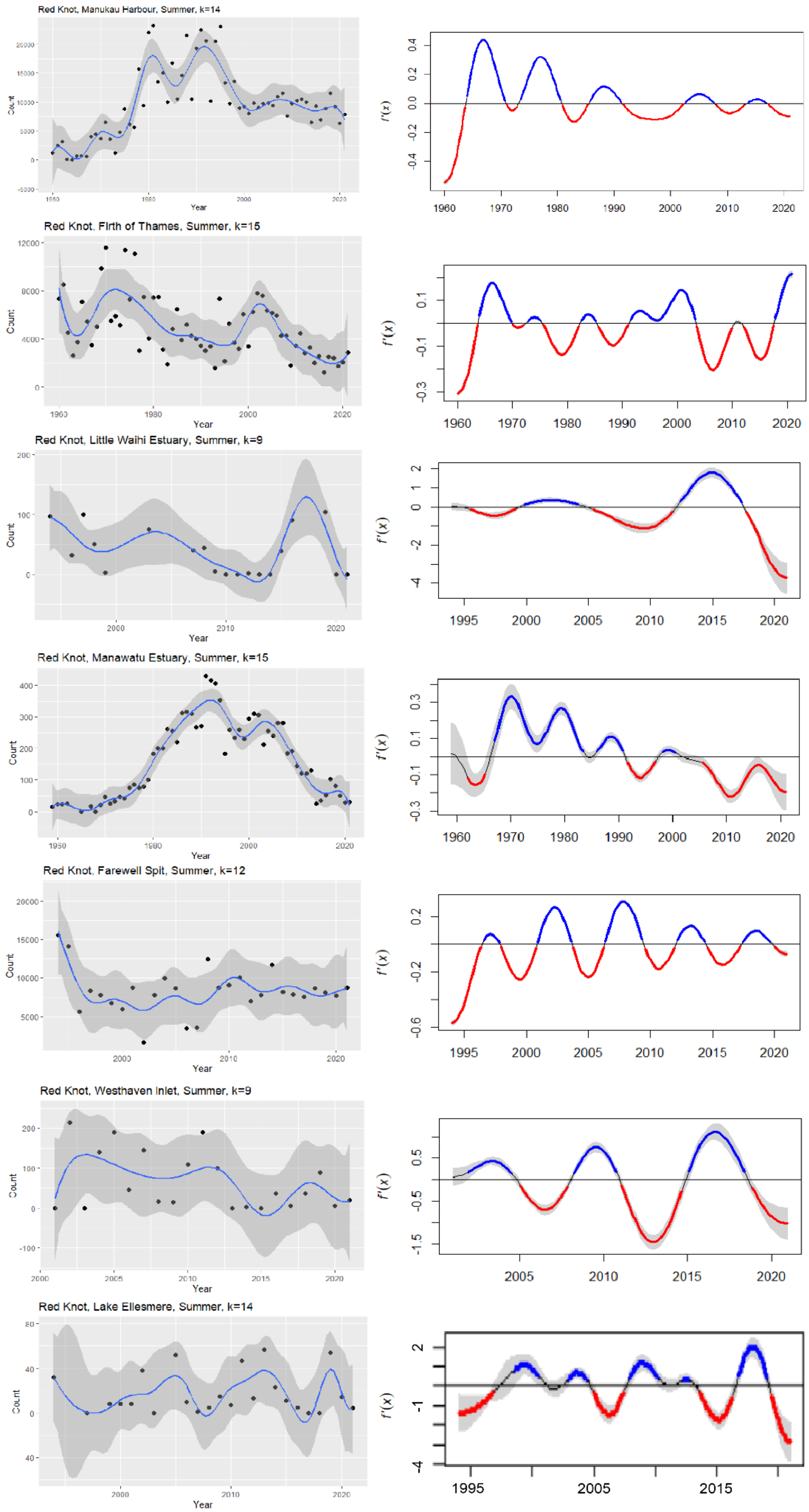


Figure 8 Continued

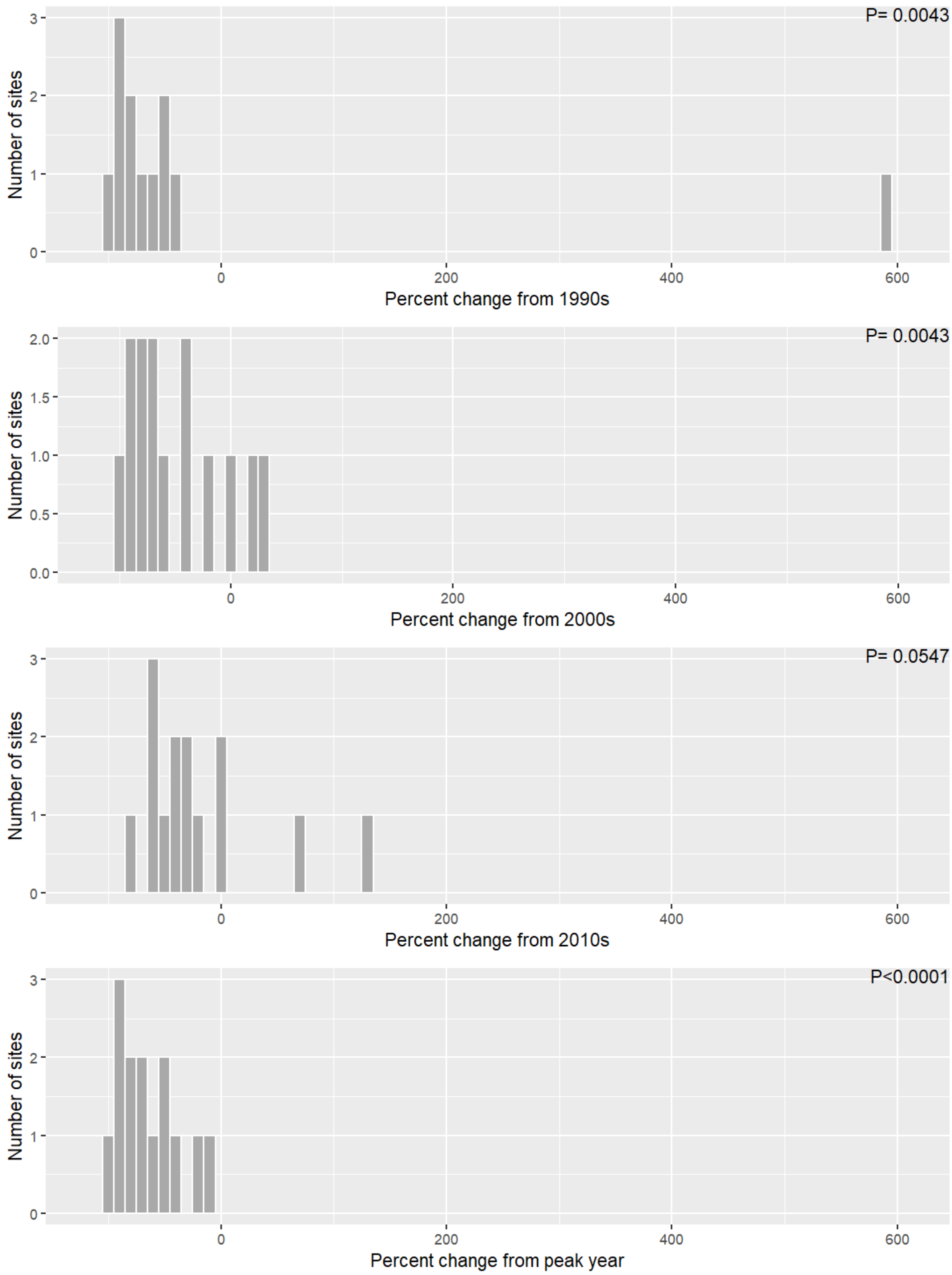


Figure 9. Individual site trends for Red Knots in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 8. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Red Knots in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Log(Area.ha)	0.36	Intercept	216.34	262.30	0.825	0.431
	$F_{1,9}$=0.7897, P=0.3973, R^2=-0.02148		Log(Area.ha)	-30.17	33.94	-0.889	0.397
2000s	Lat	0.36	Intercept	-315.002	164.941	-1.91	0.0826
	$F_{1,11}$=2.657, P=0.1314, R^2=0.1213		Latitude	7.082	4.345	1.63	0.1314
2010s	Lat	0.38	Intercept	275.583	231.352	1.191	0.259
	$F_{1,11}$=1.622, P=0.229, R^2=0.04931		Latitude	-7.763	6.095	-1.274	0.229
MaxYr	MaxCount	0.36	Intercept	-6.997e+01	9.549e+00	-7.328	1.49e-05
	$F_{1,11}$=0.7988, P=0.3906, R^2=-0.01705		MaxCount	8.964e-04	1.003e-03	0.894	0.391

Winter

Population trends for Red Knot in Winter were modelled at only four sites, but most trend values were negative, varying from -81% to an increase of 394% (Table 9, Figure 10). The median trend was a significant decline for the periods from the 1990s and the year of maximum count (median trends of -16.6% to -68.5%; significance values are given in Figure 11).

The largest population was in the Manukau Harbour, which had variable but generally increasing counts from the 1960s to the early 1990s, after which counts declined. The other sites had less obvious declines from when counts started in the mid-1990s.

Table 9. Red Knot winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Kaipara Harbour	27	8	1131	100	237.2	-81 (-97, 30)	394 (-37, 4869)	-8.2 (-85, 437)	-75 (-96, 61)	-75 (-96, 61)
Manukau Harbour	61	13	6561	1379	1995	-70 (-92, 4.4)	-41 (-86, 134)	21 (-70, 364)	-55 (-88, 79)	-49 (-88, 140)
Firth of Thames	28	10	973	136	213.6	-67 (-94, 53)	-58 (-88, 39)	-36 (-68, 19)	-67 (-94, 52)	-67 (-94, 52)
Farewell Spit	26	3	733	189.5	211.3	-48 (-82, 42)	-42 (-76, 34)	-25 (-52, 16)	-32 (-63, 23)	-48 (-81, 41)

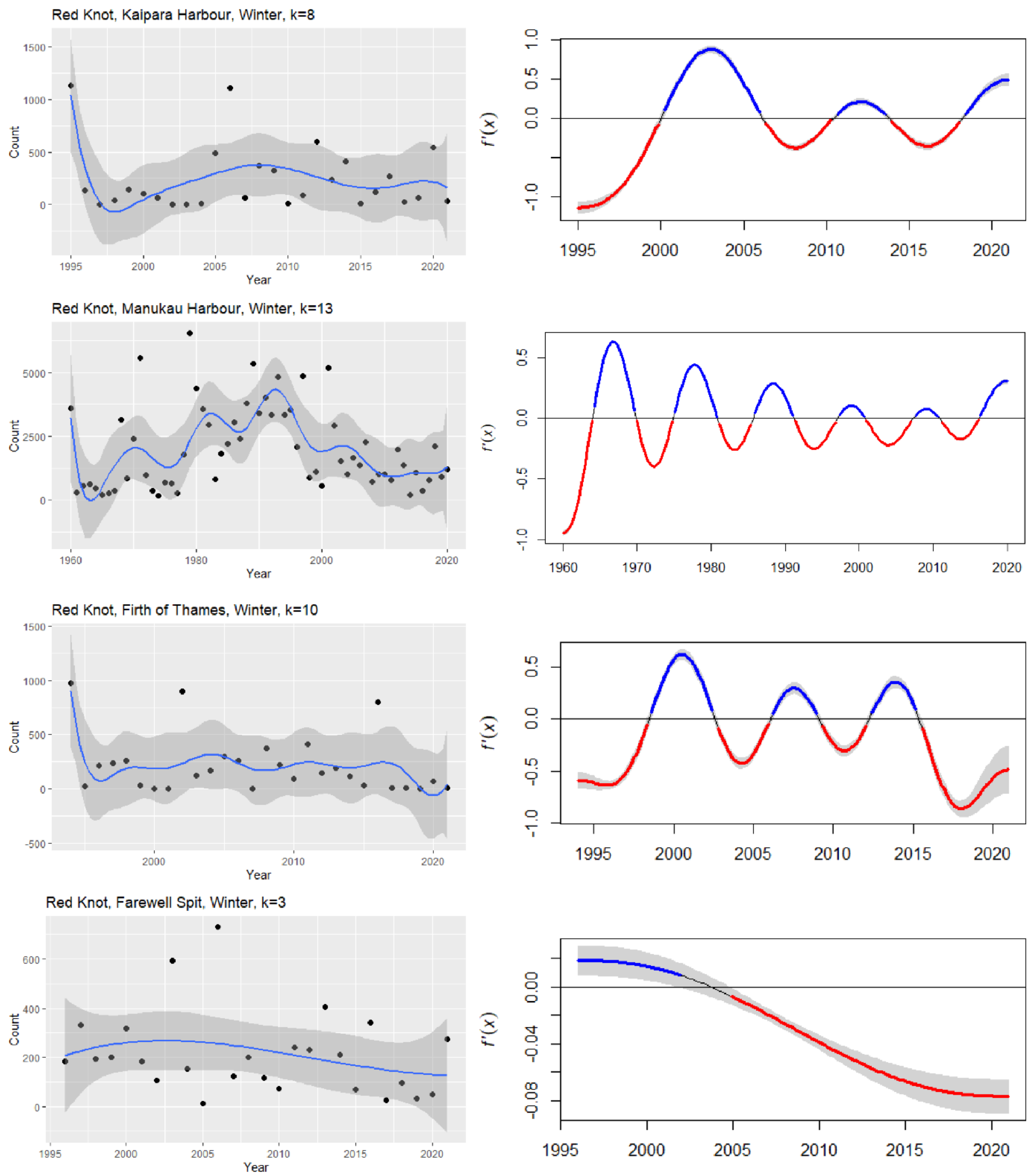


Figure 10. Site-specific trends of Red Knots in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

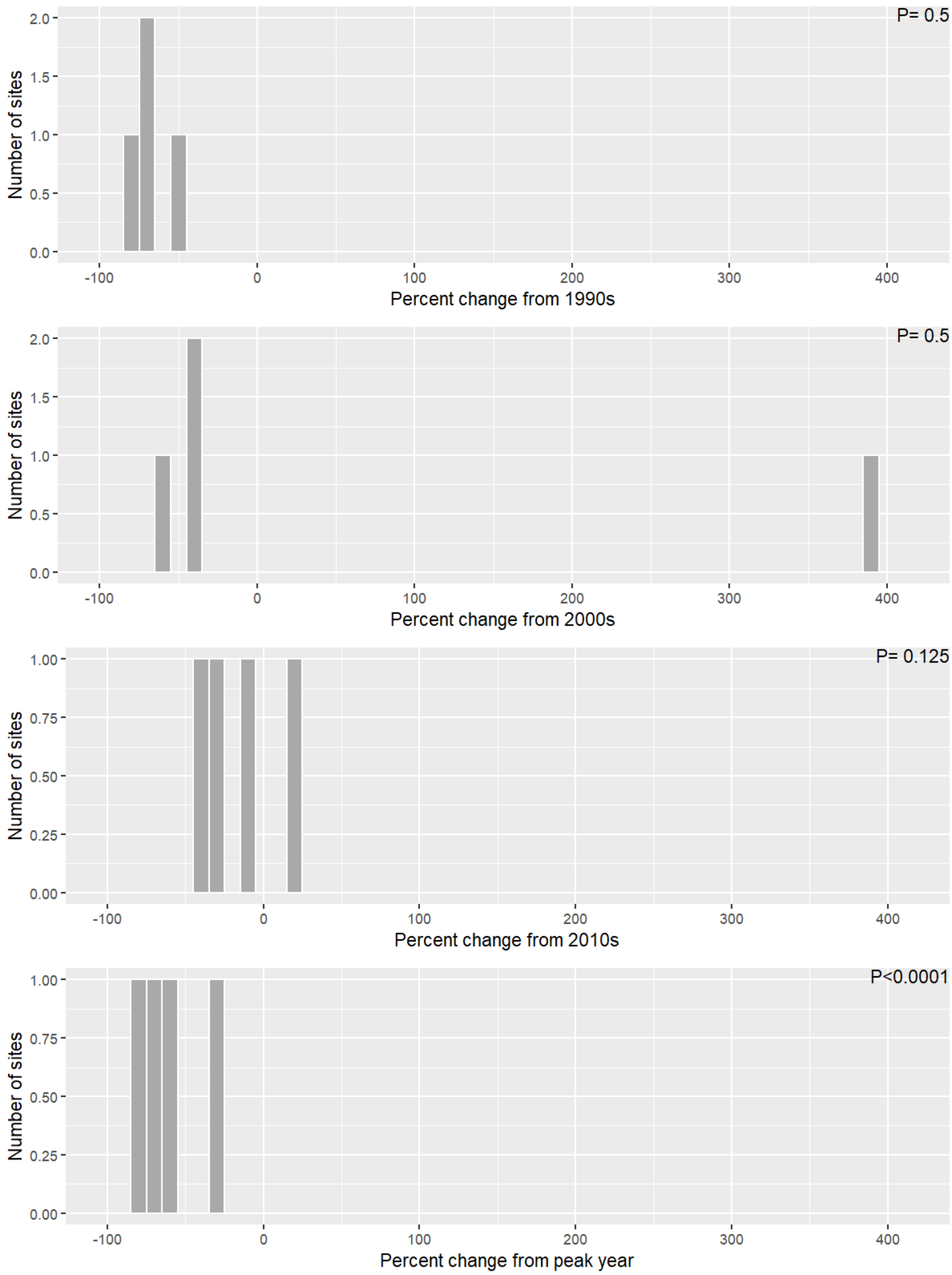


Figure 11. Individual site trends for Red Knots in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Ruddy Turnstone

Summer

Modelled population trends for Ruddy Turnstone in Summer varied from -96% to an increase of 274% depending on the site and time period (Table 10, Figure 12). The median trend was only significantly different to zero for the year of maximum count (median trends of 28.5% to -38.5%; significance values are given in Figure 13).

The common trend shown was that of the more northerly sites experiencing an increase around 2015 with six sites furthest north all experiencing a small increase in population numbers. The longer-term sites (Manukau Harbour and Firth of Thames) both experienced a significant increase in population numbers at the start of the 1980s with both sites roughly doubling in number before subsequently declining.

Models testing whether trends at individual sites varied with population size, habitat area and latitude found no significant relationships from any of the time periods studied (Table 11).

Table 10. Ruddy Turnstone summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Parengarenga Harbour	14	8	627	150	172.8	274 (-52, 3024)	140 (-95, 14416)	151 (-43, 1295)	-7 (-77, 356)	266 (-48, 2685)
Houhora Harbour	13	8	118	40	43.9	118 (-54, 762)	80 (-45, 409)	33 (-25, 118)	14 (-12, 43)	116 (-54, 744)
Rangaunu Harbour	12	7	600	225	233.6	-43 (-83, 103)	-9.8 (-72, 195)	24 (-51, 223)	-42 (-83, 106)	-42 (-83, 106)
Mangawhai Estuary	24	10	39	14.5	16.6	66 (-55, 484)	102 (-9.5, 326)	58 (-20, 230)	-24 (-65, 47)	71 (-52, 458)
Kaipara Harbour	27	14	765	301	313.2	11 (-57, 164)	-11 (-59, 88)	71 (-23, 265)	-43 (-73, 12)	0.43 (-59, 126)
Manukau Harbour	62	15	803	268.5	299	-33 (-58, 1.9)	-14 (-46, 34)	-3.5 (-37, 41)	-35 (-59, -0.3)	55 (-16, 175)
Firth of Thames	62	13	321	35	59.6	-94 (-99, -68)	-82 (-97, 18)	-60 (-92, 103)	-96 (-99, -81)	-88 (-98, -11)
Little Waihi Estuary	19	7	35	2	5.4	-85 (-98, 28)	-83 (-98, 23)	74 (-84, 1874)	-90 (-98, -24)	-85 (-98, 20)
Farewell Spit	28	15	1028	390	417.5	-72 (-90, -18)	-53 (-80, 43)	-26 (-72, 110)	-53 (-80, 25)	-68 (-88, -15)
Lake Ellesmere	24	10	66	18	19.8	13 (-68, 320)	16 (-55, 224)	16 (-38, 121)	3.5 (-11, 21)	13 (-68, 312)
Invercargill	18	3	164	50	73.5	-15 (-69, 105)	37 (-43, 181)	154 (18, 435)	-12 (-67, 110)	-12 (-67, 110)
Awarua Bay	19	6	200	67	81.9	-33 (-75, 74)	-53 (-78, -1.6)	-23 (-65, 54)	-56 (-77, -15)	-34 (-75, 68)

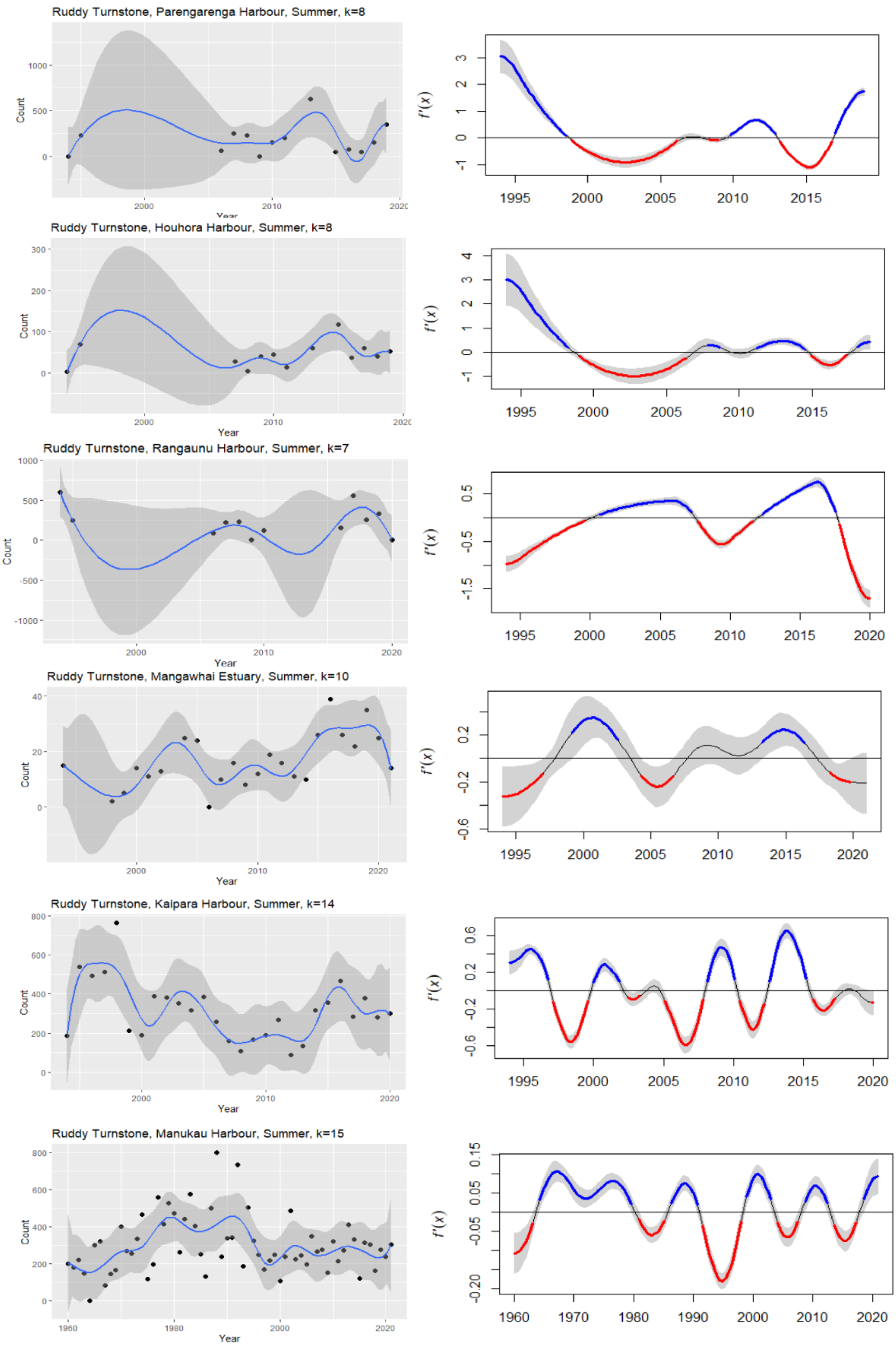
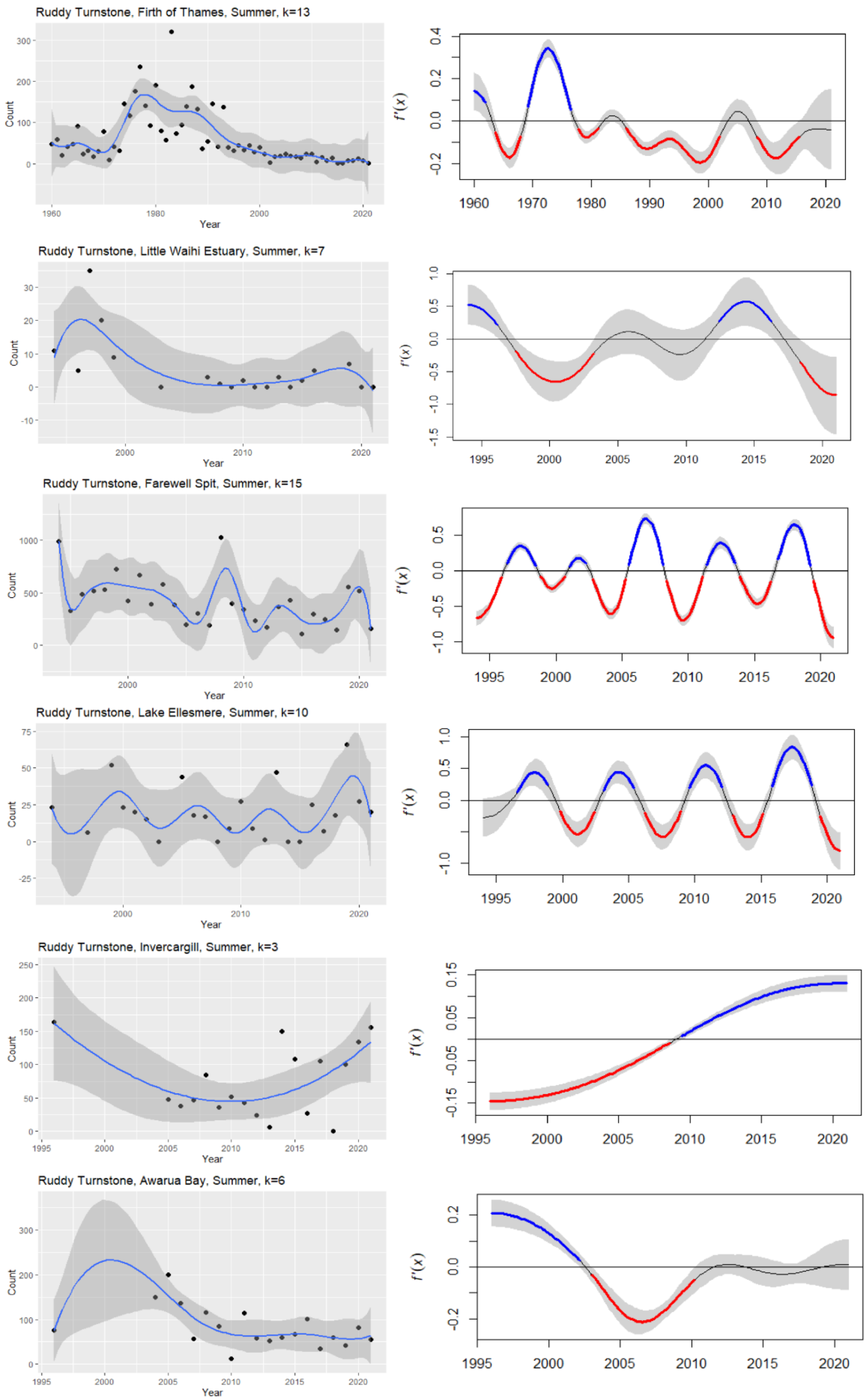


Figure 12. Site-specific trends of Ruddy Turnstones in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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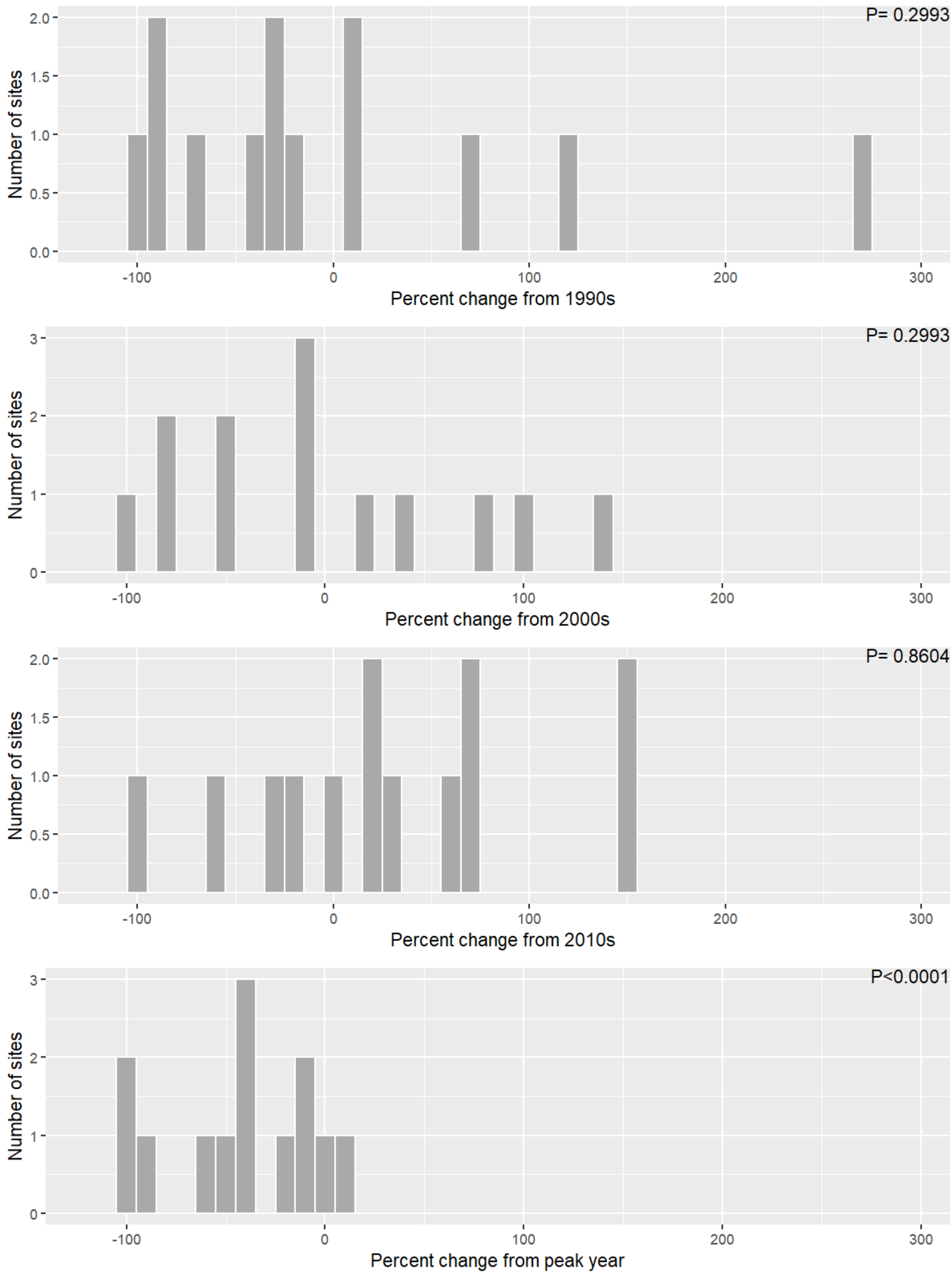


Figure 13. Individual site trends for Ruddy Turnstones in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 11. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Ruddy Turnstones in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.46	Intercept	335.977	269.169	1.248	0.24
	$F_{1,10}$=1.494, P=0.2496, R^2=0.04299		Latitude	-8.419	6.888	-1.222	0.25
2000s	Lat	0.36	Intercept	190.963	192.054	0.994	0.344
	$F_{1,10}$=0.941, P=0.3549, R^2=-0.005393		Latitude	-4.767	4.914	-0.970	0.355
2010s	Log(Area.ha)	0.32	Intercept	98.468	108.593	0.907	0.386
	$F_{1,10}$=0.3098, P=0.59, R^2=-0.06694		Log(Area. ha)	-7.661	13.762	-0.557	0.590
MaxYr	MaxCount	0.32	Intercept	-31.86756	15.99083	-1.993	0.0743
	$F_{1,10}$=0.1564, P=0.7008, R^2=-0.08306		MaxCount	-0.01219	0.03082	-0.395	0.7008

Winter

Population trends for Ruddy Turnstone in Winter varied were modelled for only two sites. Trends varied from -3.6% to 391% depending on the site and time period (Table 12, Figure 14, Figure 15). The long-term trend at the Manukau Harbour indicates a period of higher counts in the 1980s followed by a decrease, with smaller peaks around 2010 and 2020. The smaller population at Awarua Bay in Southland shows a slight increase in recent years.

Table 12. Ruddy Turnstone winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Manukau Harbour	61	14	292	51	64.4	0.083 (-59, 117)	45 (-46, 260)	36 (-43, 210)	-3.6 (-61, 128)	141 (-28, 686)
Awarua Bay	19	3	38	9	11.3	391 (77, 1247)	276 (61, 770)	100 (28, 211)	0 (0, 0)	384 (76, 1213)

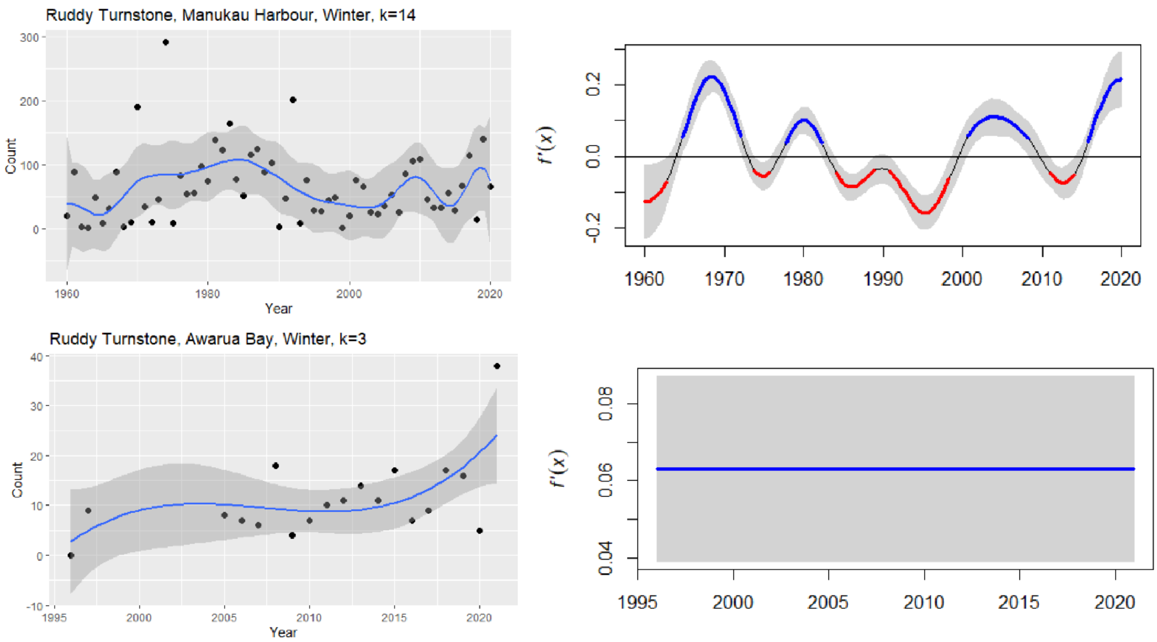


Figure 14. Site-specific trends of Ruddy Turnstones in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

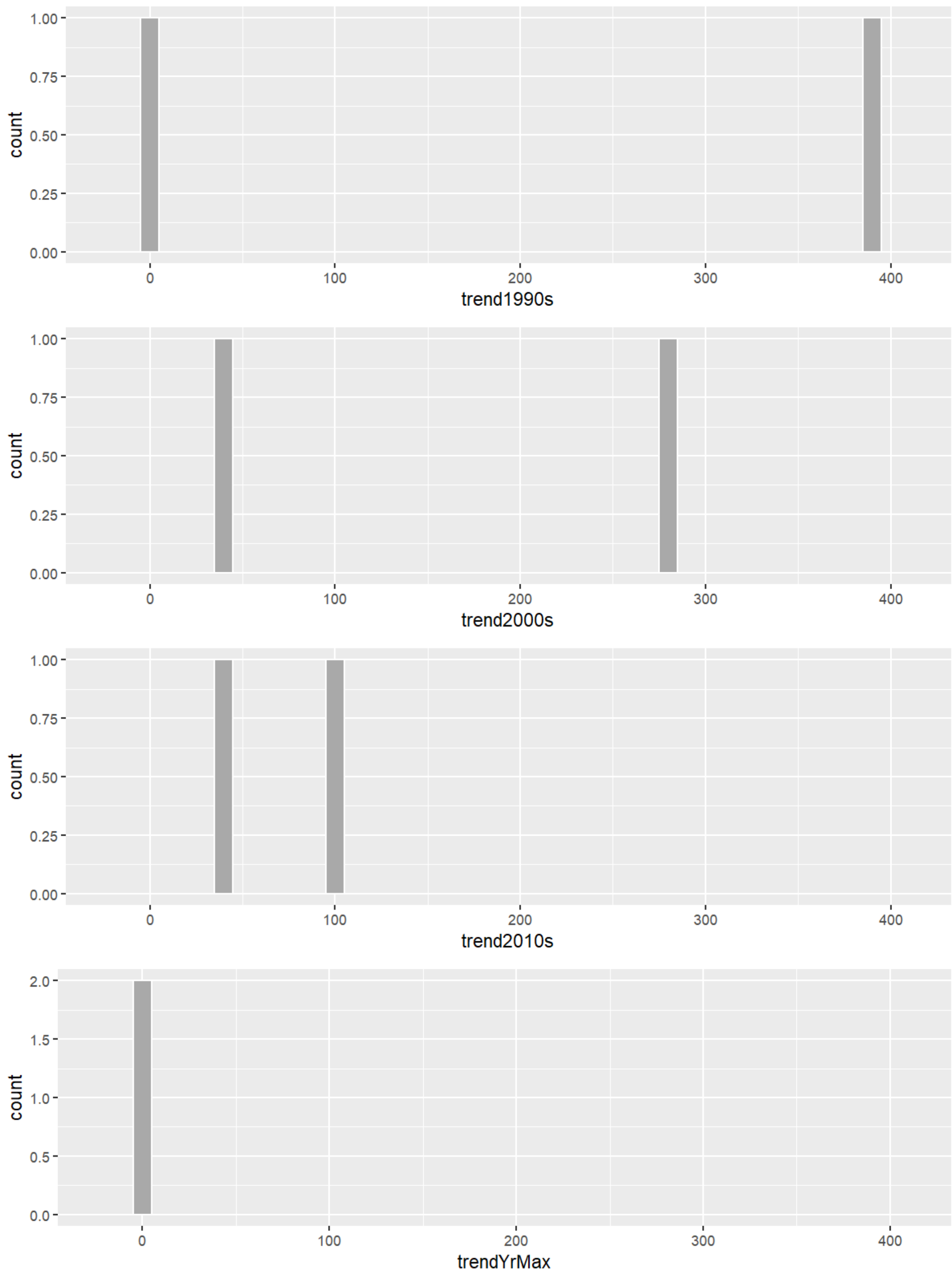


Figure 15. Individual site trends for Ruddy Turnstones in winter over different analysis periods.

Pacific Golden Plover

Summer

Modelled population trends for Pacific Golden Plover in summer varied from -98% to an increase of 259% depending on the site and time period (Table 13, Figure 16). The median trend was negative and significantly different to zero, or almost so, for most periods of the study, (median trends of -28.5% to -60.5%; significance values are given in Figure 17).

The three longer-term sites (Manukau Harbour, Firth of Thames and Manawatū Estuary) showed a decline in the early 1980s which remained a constant decline for both the Manawatū Estuary and the Firth of Thames while the Manukau Harbour showed a temporary resurgence of the population in the early 1990s. It since declined to the level present in the 1980s. Other sites such as Ohiwa Harbour and Lake Wairarapa have shown a near constant decline. Kaipara Harbour does show some positive signs with stable counts in the 40s for the last four years.

This resurgence at the larger sites is reinforced by the results of linear models testing whether trends at individual sites varied with population size, habitat area and latitude (Table 14). Early trends from the 1990s and 2000s were positively related to population size, meaning that larger sites had more positive (or less negative) trends than sites with smaller populations; this indicates that sites with larger populations are declining slower than those with smaller populations. Trends from the 2010s and the first year of counts were positively related to area (higher trends at larger sites), with latitude also being significant from the first year (positively, so more southern sites had higher trends).

Table 13. Pacific Golden Plover summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Houhora Harbour	13	3	17	4	5.1	-63 (-90, 34)	-61 (-88, 24)	-47 (-79, 32)	-60 (-88, 27)	-63 (-90, 31)
Kaipara Harbour	27	14	100	29	33.2		-28 (-74, 156)	34 (-46, 259)	16 (-52, 210)	-26 (-74, 157)
Manukau Harbour	62	15	118	21.5	27.5	202 (-81, 4197)	194 (-77, 5494)	-42 (-94, 318)	-87 (-98, 3.2)	-86 (-98, 14)
Firth of Thames	62	15	246	9	34.1		-48 (-90, 144)	-39 (-82, 94)	-48 (-90, 142)	-48 (-90, 142)
Little Waihi Estuary	19	12	91	18	19.3		259 (-56, 2827)	139 (-43, 897)	37 (-19, 131)	255 (-56, 2731)
Ohiwa Harbour	26	3	18	8	7.7	173 (-32, 1100)	-24 (-73, 86)	91 (-33, 505)	-10 (-68, 127)	152 (-35, 911)
Ahuriri Estuary	25	13	27	8	8	-32 (-91, 385)	-50 (-89, 120)	-17 (-84, 244)	-65 (-92, 61)	-33 (-91, 331)
Manawatū Estuary	60	14	53	17	19.1			-55 (-71, -33)	-55 (-71, -33)	-55 (-71, -33)

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Farewell Spit	28	11	37	5.5	9.5	-98 (-99, -94)	-97 (-99, -91)	-96 (-98, -88)	-98 (-99, -95)	-97 (-99, -92)
Lake Wairarapa	12	6	27	16	15.1		-65 (-97, 549)	-26 (-96, 1435)	-79 (-98, 279)	-73 (-98, 513)
Lake Ellesmere	24	12	80	28.5	28.5	-58 (-88, 24)	95 (-47, 673)	107 (-56, 823)	-55 (-85, 52)	-28 (-82, 249)
Invercargill	18	5	69	3.5	15	-65 (-83, -31)	-51 (-69, -22)	-31 (-46, -12)	-69 (-86, -34)	-69 (-86, -34)

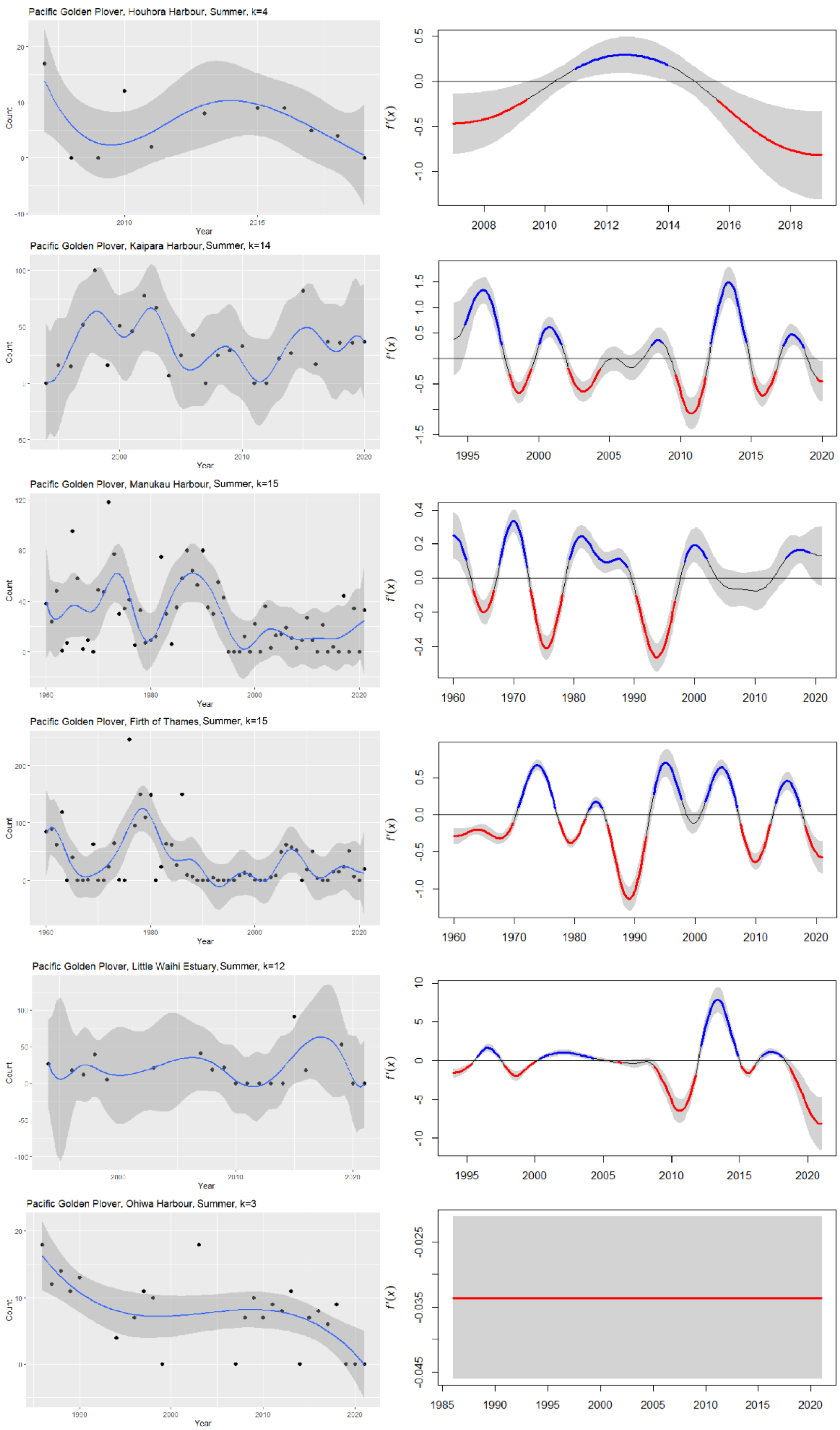


Figure 16. Site-specific trends of Pacific Golden Plovers in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

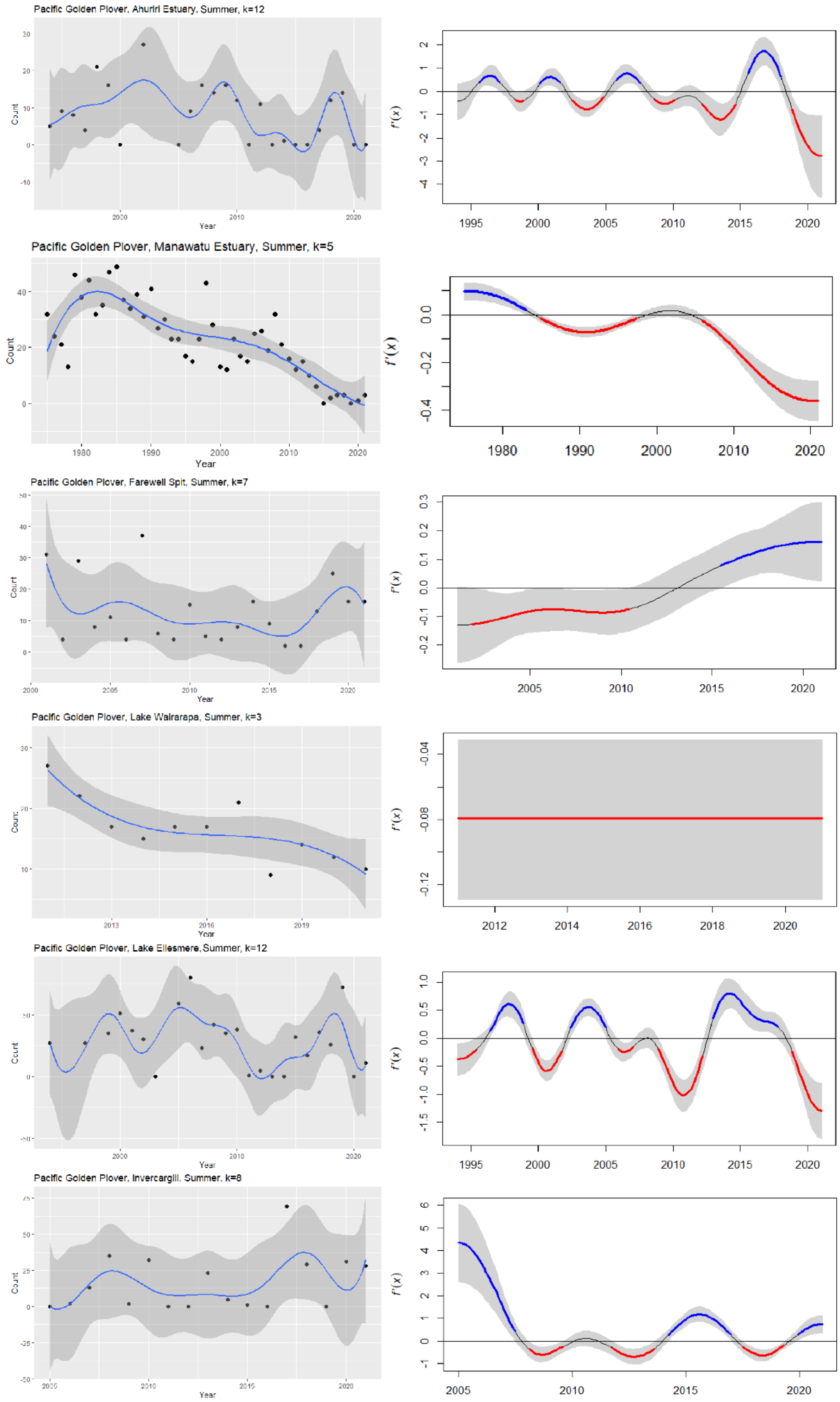


Figure 16 Continued

Table 14. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Pacific Golden Plovers in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	MaxCount	0.77	Intercept	-114.1848	50.6914	-2.253	0.0652
	$F_{1,6}$=7.58, P=0.03316, R^2=0.4845		MaxCount	1.2325	0.4477	2.753	0.0332
2000s	MaxCount	0.51	Intercept	-71.0682	47.9656	-1.482	0.1726
	$F_{1,9}$=4.867, P=0.05478, R^2=0.2789		MaxCount	1.0631	0.4819	2.206	0.0548
2010s	Log(Area.ha)	0.56	Intercept	-180.893	55.830	-3.240	0.00887
	$F_{1,10}$=11.53, P=0.006822, R^2=0.4891		Log(Area. ha	25.715	7.573	3.396	0.00682
MaxYr	Log(Area.ha)	0.28	Intercept	-111.774	40.698	-2.746	0.0206
	$F_{1,10}$=2.674, P=0.1331, R^2=0.132		Log(Area. ha)	9.027	5.521	1.635	0.1331

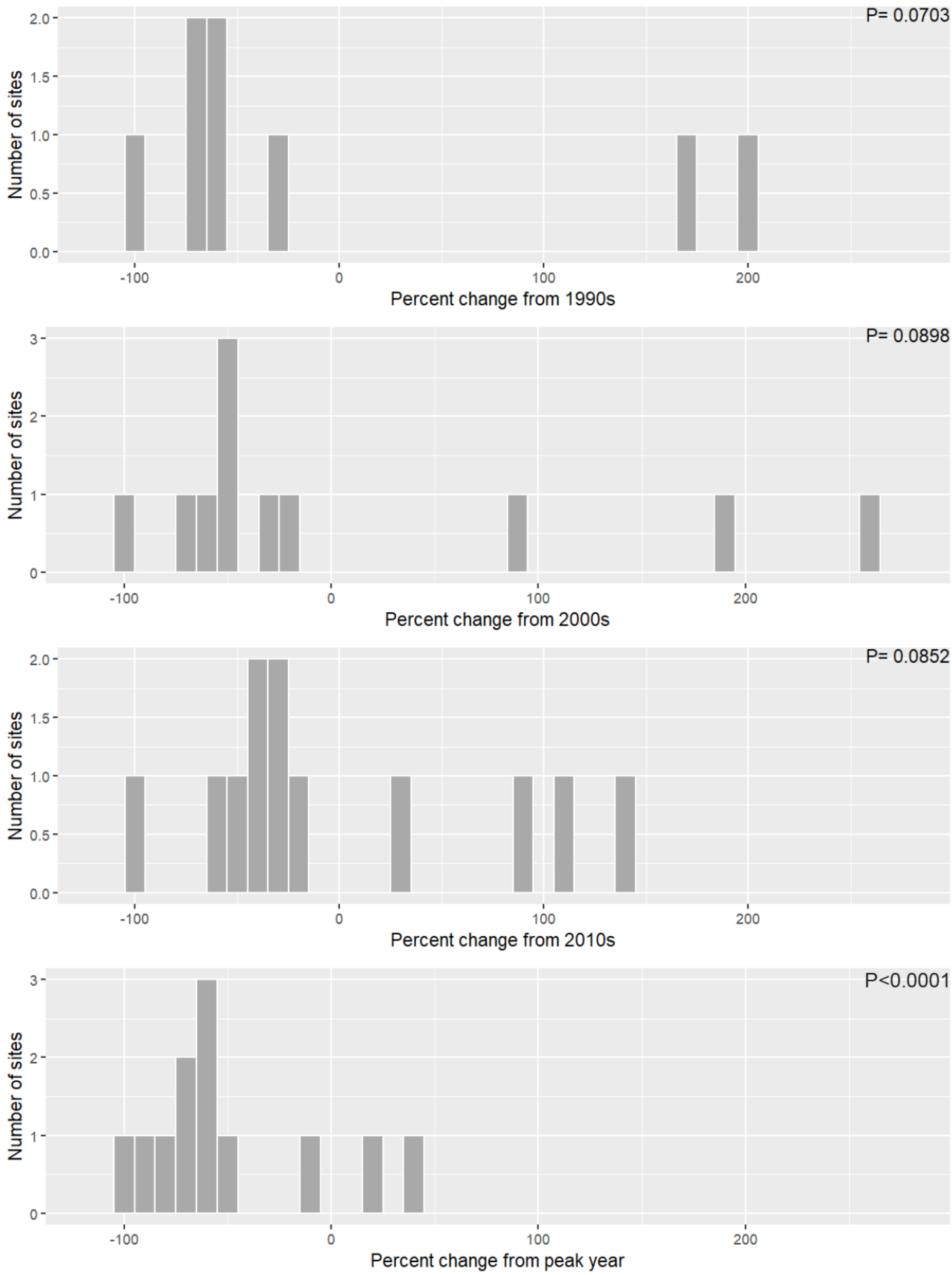


Figure 17. Individual site trends for Pacific Golden Plovers in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Red-necked Stint

Summer

Red-necked Stints are relatively uncommon birds at most sites, and only six sites had sufficient numbers for analysis. Modelled population trends for stints in Summer varied from -89% to an increase of 492% depending on the site and time period, but most trends were negative (Table 15, Figure 18). The median trend was significantly lower than zero most time periods studied (median trends of -63.5% to -72.5%; significance values are given in Figure 19).

The long-term data series for the Manukau Harbour reveal periods of increase to the mid-1980s and the mid-2000s (Figure 18). At the Firth of Thames, numbers declined fairly consistently from the mid-1960s. At Lake Ellesmere, the stronghold of the species in New Zealand, numbers declined steadily from 2000 until a high count again in 2019.

Table 15. Red-necked Stint summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Kaipara Harbour	27	9	21	6	6.6	-89 (-98, -18)	-85 (-97, -4.4)	-69 (-95, 124)	-88 (-98, -20)	-88 (-98, -18)
Manukau Harbour	62	10	35	11	12.2	-62 (-87, 16)	-78 (-92, -34)	-69 (-91, -4.1)	-81 (-94, -43)	89 (-60, 742)
Firth of Thames	62	15	20	3	4.5	-71 (-93, 77)	-61 (-91, 85)	-61 (-93, 117)	-86 (-97, -31)	-71 (-94, 50)
Farewell Spit	28	10	26	3.5	5.1	492 (-53, 7020)	7.6 (-77, 329)	-66 (-92, 19)	-64 (-91, 33)	449 (-53, 6179)
Lake Ellesmere	24	13	112	46.5	52.5	-29 (-70, 73)	-64 (-80, -24)	86 (-7.7, 338)	-52 (-76, 18)	-30 (-69, 63)
Awarua Bay	19	8	51	20	21.9	-76 (-94, 19)	-80 (-94, -30)	-41 (-83, 124)	-45 (-82, 87)	-76 (-94, 16)

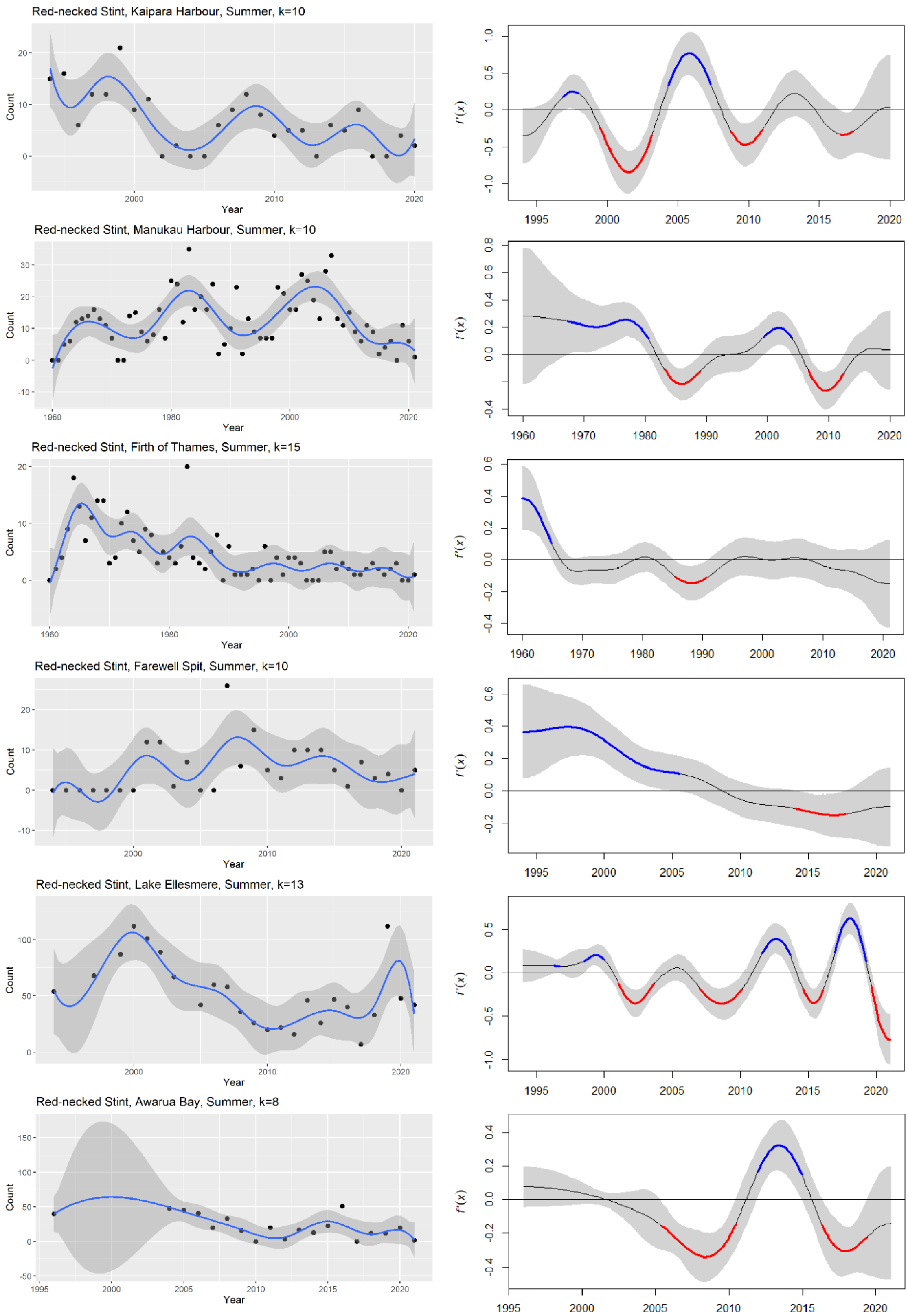


Figure 18. Site-specific trends of Red-necked Stints in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

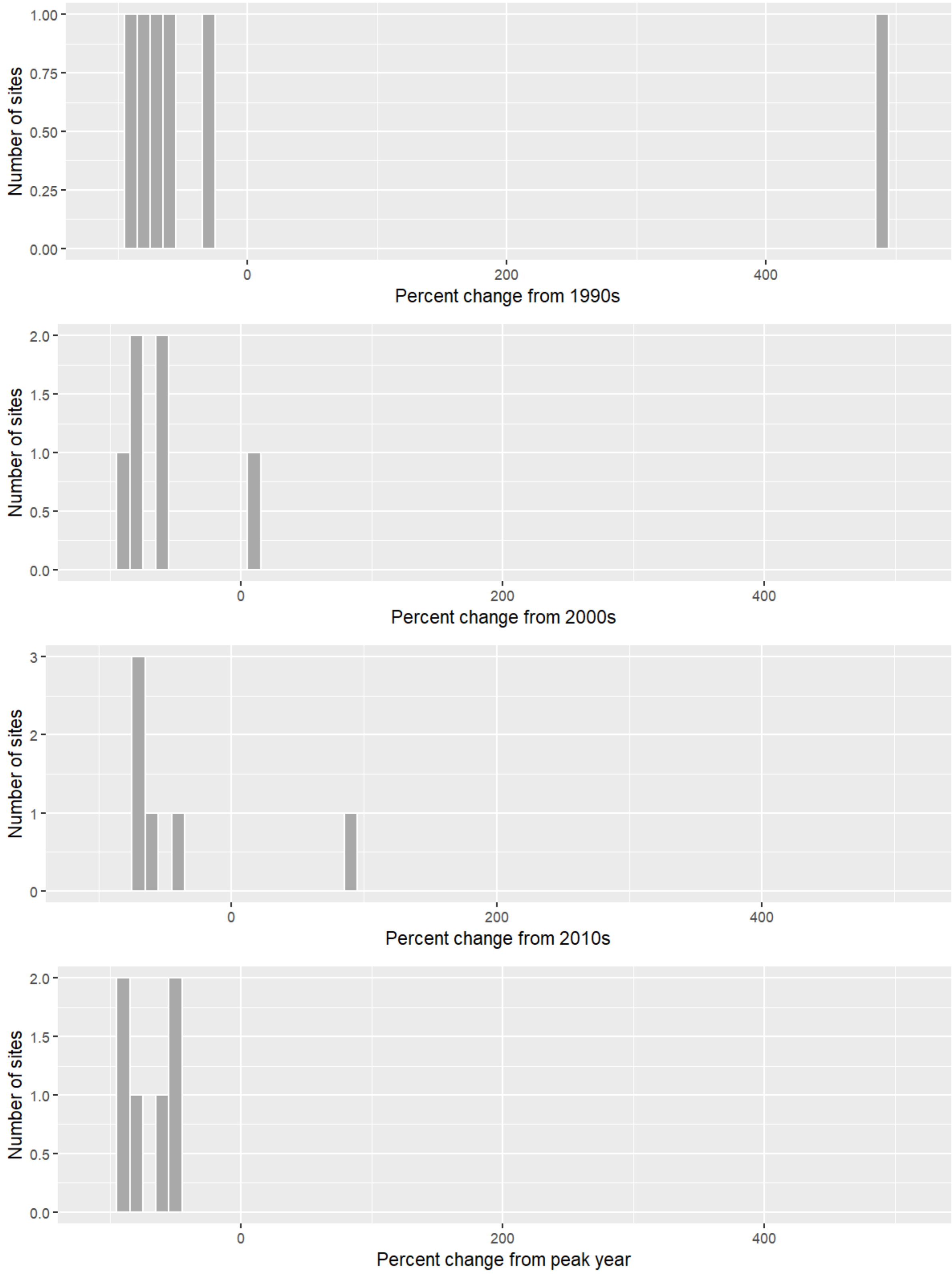


Figure 19. Individual site trends for Red-necked Stints in summer over different analysis periods.

Winter

Red-necked Stints were analysed for only two sites in winter. Modelled population trends for Red-necked Stint in Winter varied from -86% to an increase of 1578% depending on the site and time period (Table 16, Figure 20).

The two sites showed no common trend with Lake Ellesmere experiencing a near constant decline since counts started, while Manukau Harbour experienced periods of increase from the mid-1970s to the mid-1980s and again around the mid-2000s.

Table 16. Red-necked Stint winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Manukau Harbour	61	15	24	1	3.7	-58 (-94, 228)	-63 (-95, 149)	-62 (-95, 172)	-72 (-97, 102)	1578 (-82, 170786)
Lake Ellesmere	23	14	27	5	8.1	-86 (-96, -57)	-80 (-92, -49)	-57 (-73, -29)	-80 (-92, -49)	-86 (-96, -56)

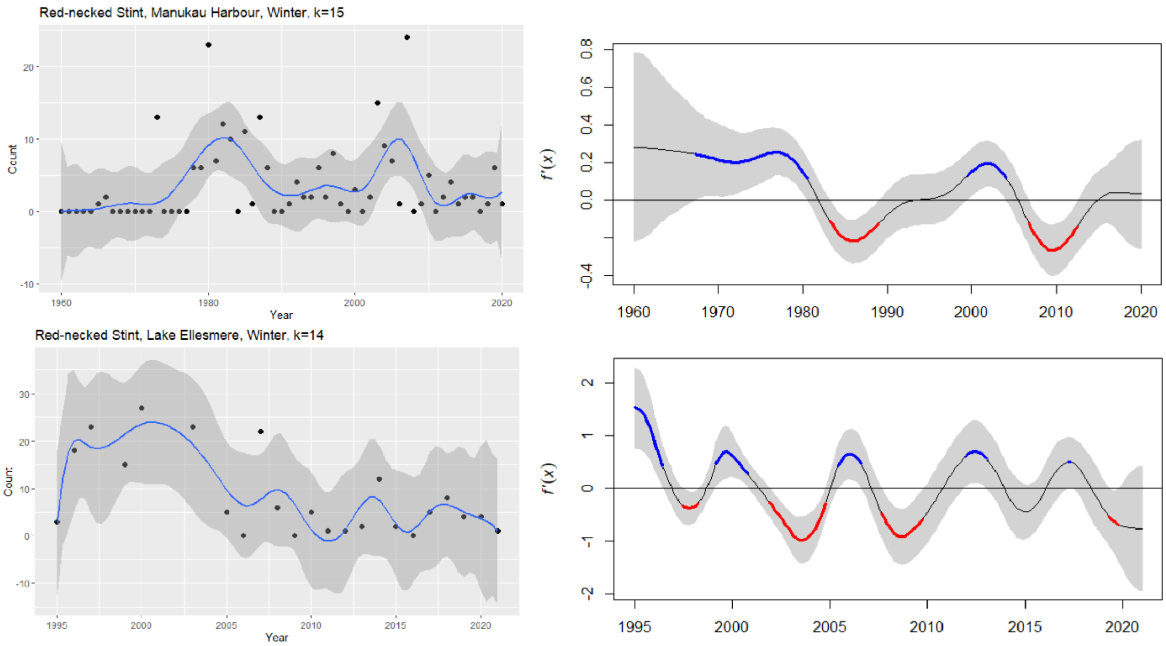


Figure 20. Site-specific trends of Red-necked Stints in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

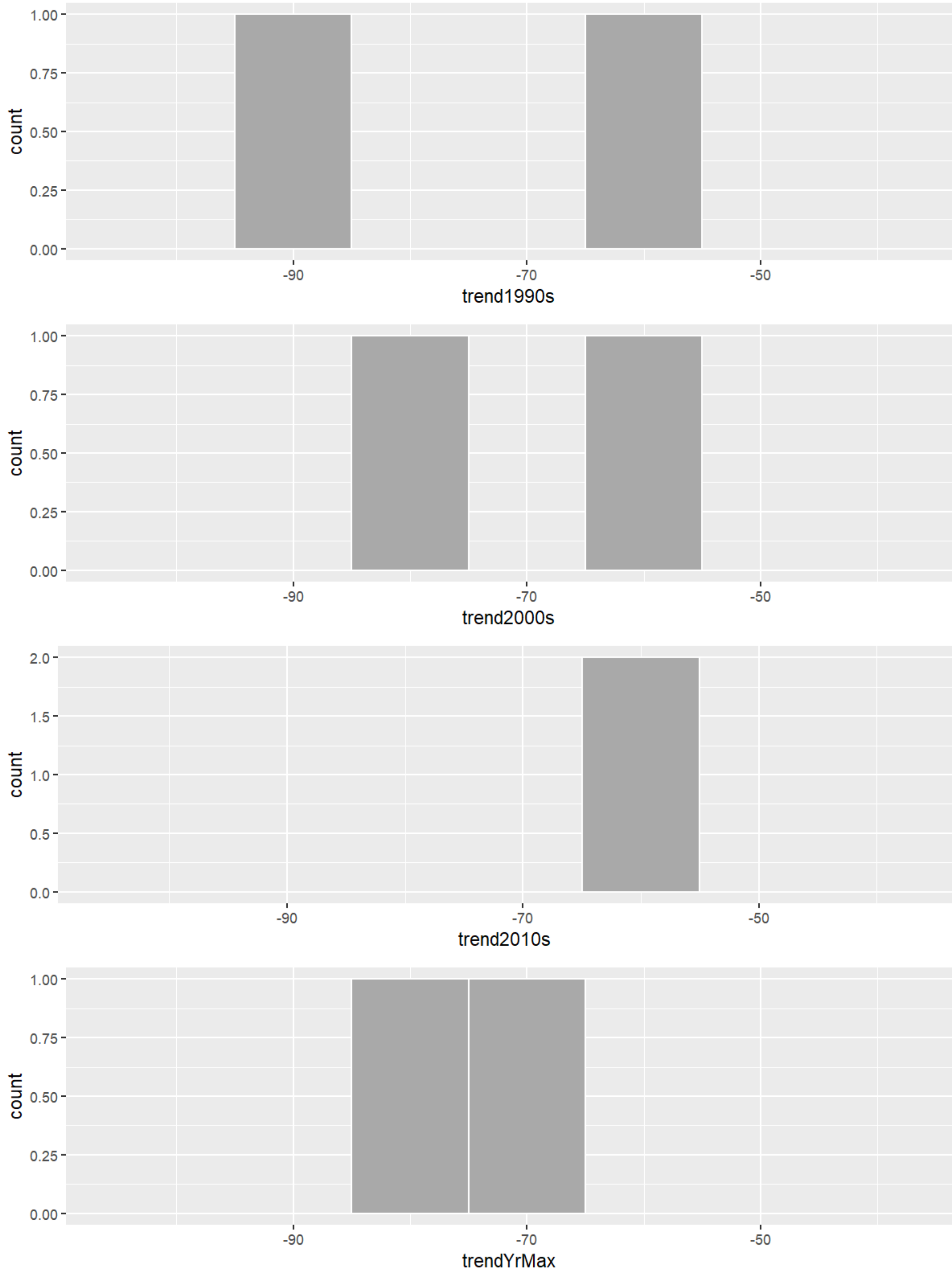


Figure 21. Individual site trends for Red-necked Stints in winter over different analysis periods.

Sharp-tailed Sandpiper

Summer

While population trends for Sharp-tailed Sandpiper in Summer were modelled at only three sites, all were negative, varying from -96% to -31% depending on the site and time period (Table 17, Figure 22). The median trend was significantly different to zero for all periods of the study (median trends of -71% to -96%; significance values are given in Figure 23).

All three sites had highest numbers in the 1980s. They declined gradually at the Manawatū Estuary but more steeply at the Manukau Harbour and Firth of Thames. Sharp-tailed Sandpipers were barely recorded in summer censuses at the Manukau Harbour and Manawatū Estuary in the 2010s.

Table 17. Sharp-tailed Sandpiper summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Manukau Harbour	62	14	18	2	4	-75 (-96, 68)	-31 (-89, 372)	-59 (-93, 179)	-88 (-98, -28)	-57 (-95, 243)
Firth of Thames	62	15	40	4	6.4	-96 (-99, -85)	-93 (-97, -78)	-78 (-88, -59)	-96 (-99, -85)	-87 (-96, -56)
Manawatū Estuary	60	3	23	3	4.8	-95 (-100, 308)	-93 (-100, 494)	-71 (-100, 2305)	-97 (-100, 112)	-68 (-100, 3047)

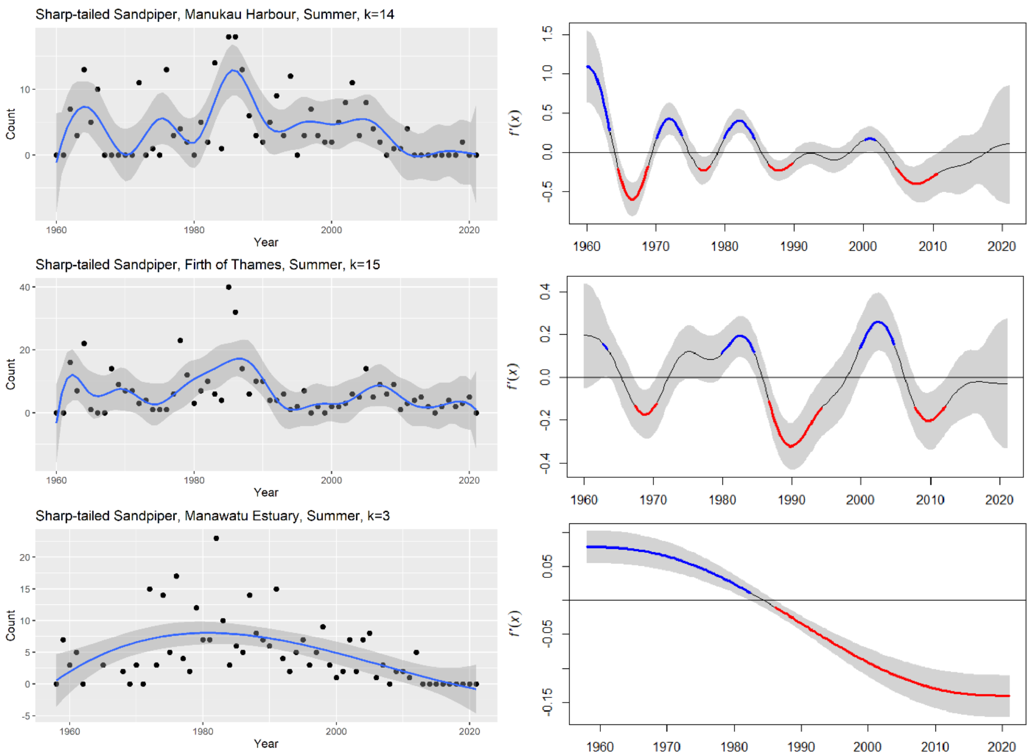


Figure 22. Site-specific trends of Sharp-tailed Sandpipers in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

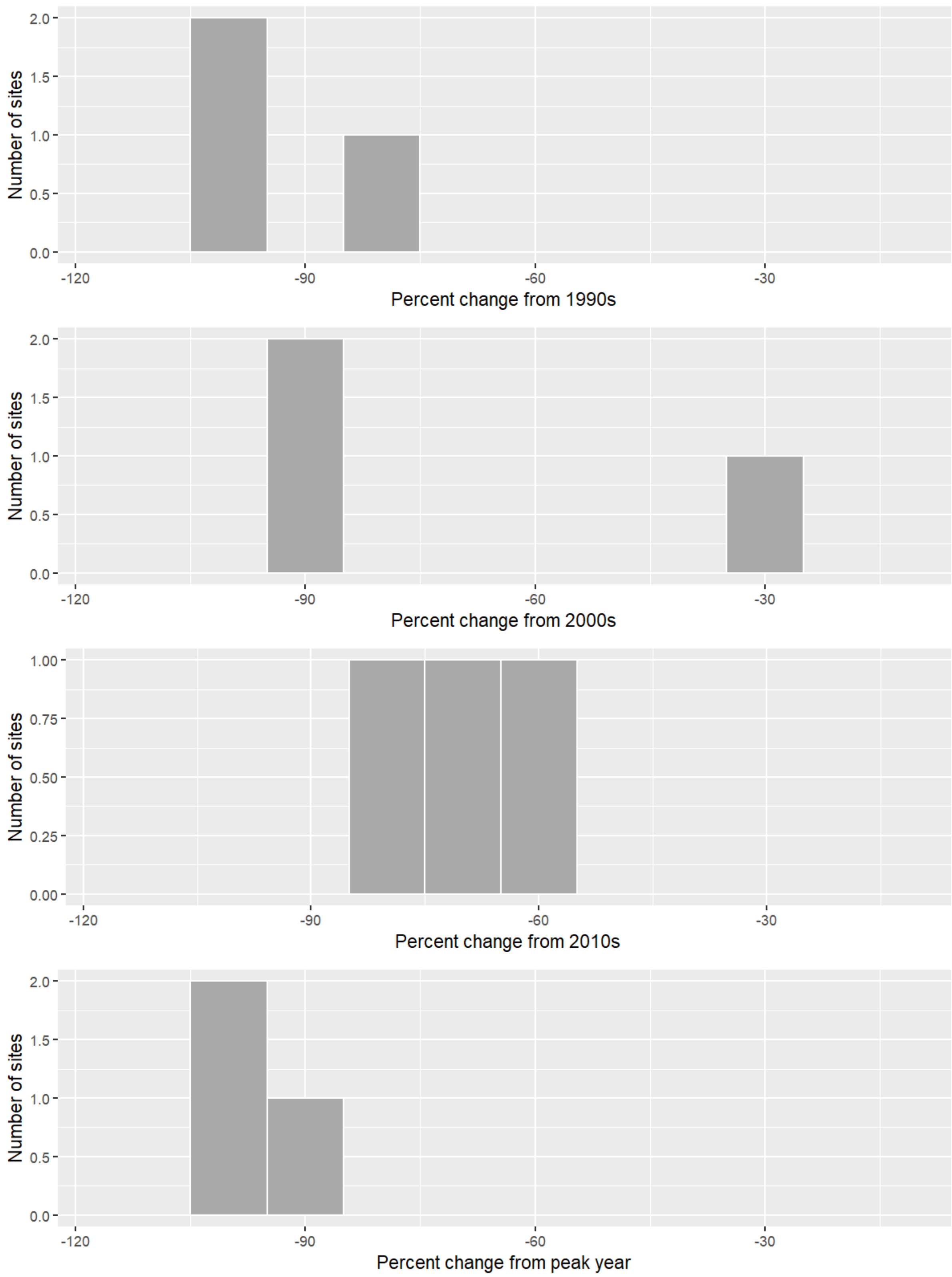


Figure 23. Individual site trends for Sharp-tailed Sandpipers in summer over different analysis periods.

Internal Migrants

South Island Pied Oystercatcher

Winter

Modelled population trends for SIPO in winter varied from -84% to an increase of 676% depending on the site and time period (Table 18, Figure 24). The median trend was not different to zero for the periods from the 1990s, 2000s, 2010s or from the first year of survey (median trends of -15.1% to 6.15%; significance values are given in Figure 25), but it was significantly lower than zero for the trends from the year of maximum count (-29%, P=0.0026).

The long-term trend at the Manukau Harbour indicates the population grew greatly from the 1960s to the 1990s but decreased thereafter (Figure 24). A post-1990s decline is evident at some sites (e.g., Whangarei, Firth of Thames) but not at others, and some sites have increased over the same period. This variability is reflected in the results of the linear models (Table 19). Latitude was negatively related to trend in two models, meaning that southern sites tended to have lower trends; the models additionally found negative trends with maximum count in one model (larger populations experienced lower trends) and with area in the other (larger sites had lower trends). The large northern sites such as Whangarei, Kaipara and Manukau Harbours, and Farewell Spit, have all shown substantial declines; likewise, sites in the southern half of New Zealand (Manawatū Estuary to Otago) have also tended to decrease. Sites in between—the Waikato and Bay of Plenty harbours and estuaries—have been generally stable or increasing.

Table 18. South Island Pied Oystercatcher winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	13	2197	808	946.08	-42 (-72, 15)	-0.059 (-52, 92)	95 (-8, 306)	-0.059 (-52, 92)	-41 (-71, 15)
Mangawhai Estuary	23	14	212	27	53.5	73 (-78, 1582)	-12 (-80, 270)	102 (-64, 1003)	-28 (-83, 237)	51 (-75, 1045)
Kaipara Harbour	27	14	28197	16817	16852.8	-26 (-45, 1.5)	-26 (-42, -2.7)	-21 (-36, -0.97)	-26 (-44, -2.7)	-26 (-45, 1.3)
Manukau Harbour	61	15	37696	22345	20631.4	-22 (-39, -1.9)	-30 (-46, -12)	-15 (-34, 7.5)	-30 (-45, -12)	676 (387, 1130)
Firth of Thames	28	15	29274	10078	11554.7	-66 (-79, -49)	-50 (-68, -26)	-14 (-46, 28)	-56 (-72, -34)	-65 (-78, -48)
Tauranga Harbour	20	12	3087	1799.5	1715.7	34 (-44, 183)	31 (-34, 161)	-5.9 (-50, 82)	-21 (-58, 36)	30 (-42, 174)
Little Waihi Estuary	19	10	578	204	215.8	111 (-22, 406)	81 (-18, 266)	37 (-9.9, 97)	0 (0, 0)	109 (-22, 398)
Ohiwa Harbour	24	13	1772	479.5	524.3	354 (157, 693)	107 (22, 251)	35 (-15, 115)	8.7 (-14, 37)	553 (238, 1182)
Aotea Harbour	24	13	2007	795	878.6	-8.2 (-63, 130)	-8.8 (-58, 100)	-28 (-65, 61)	-28 (-68, 59)	-7.7 (-63, 126)
Kawhia Harbour	25	11	5233	2467	2591.6	88 (-7.7, 310)	18 (-29, 106)	-2.1 (-42, 68)	-2.1 (-42, 68)	81 (-8.3, 283)
Manawatū Estuary	59	15	204	52	59.2	29 (-34, 174)	39 (-35, 185)	4.4 (-45, 112)	-41 (-68, 17)	89 (-25, 400)
Farewell Spit	26	14	10595	5937.5	5760.6	-63 (-80, -34)	-60 (-76, -34)	-45 (-68, -10)	-60 (-76, -34)	-63 (-79, -35)
Golden Bay	26	12	7958	2715	3319.7	114 (-9.5, 376)	91 (6.5, 246)	-3 (-40, 68)	-56 (-73, -28)	118 (-2.4, 346)
Avon-Heathcote Estuary	23	12	4726	2598	2671.2	-52 (-69, -24)	-51 (-67, -26)	-41 (-60, -13)	-65 (-76, -49)	-53 (-70, -28)
Lake Ellesmere	23	13	190	48	60.8	-65 (-96, 133)	-55 (-93, 177)	-83 (-97, -17)	-84 (-97, -27)	-61 (-94, 147)
Otago Harbour	18	9	1147	418	465.4	-72 (-82, -53)	-63 (-75, -45)	-41 (-52, -27)	-71 (-82, -52)	-71 (-82, -52)

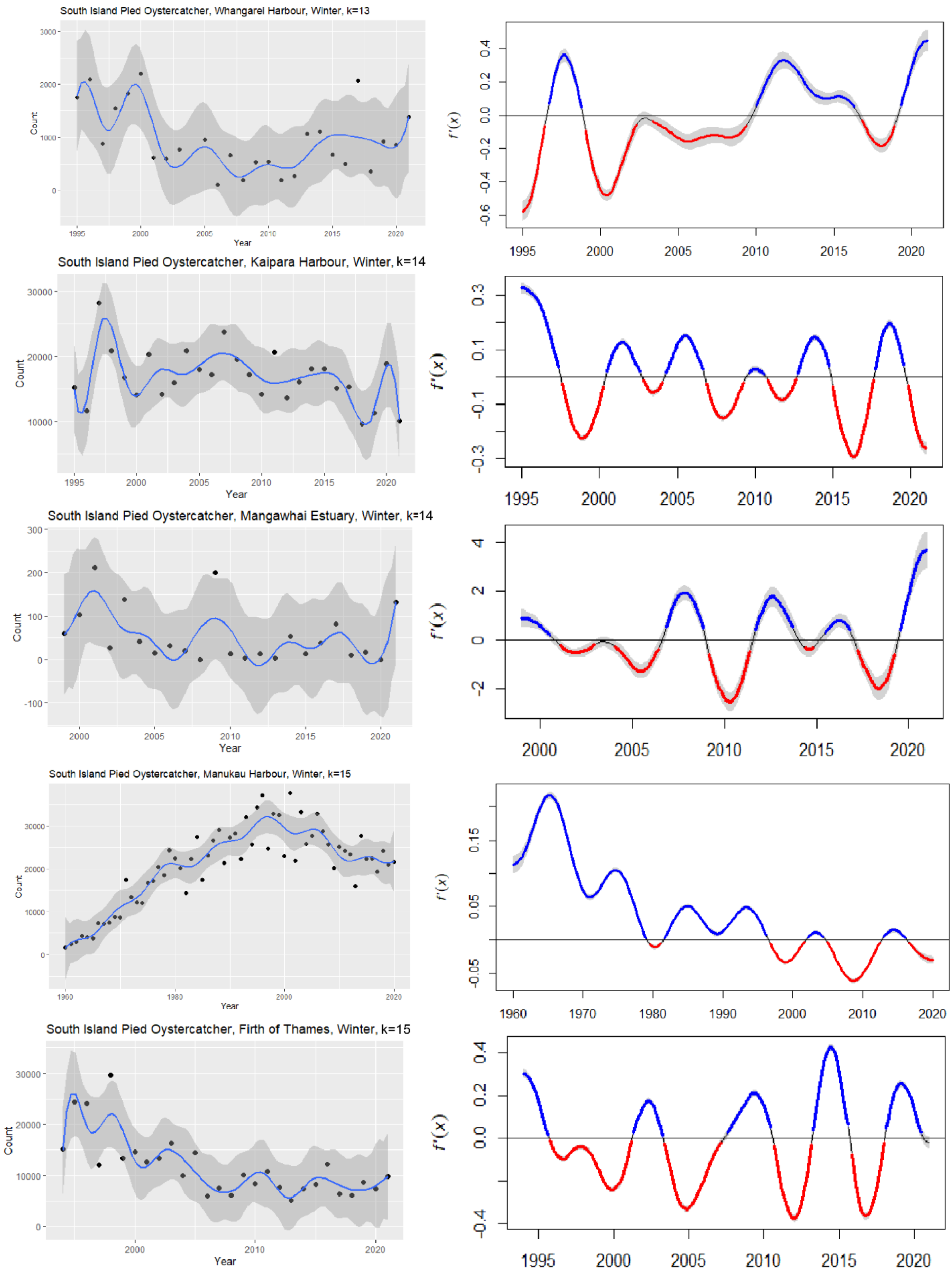
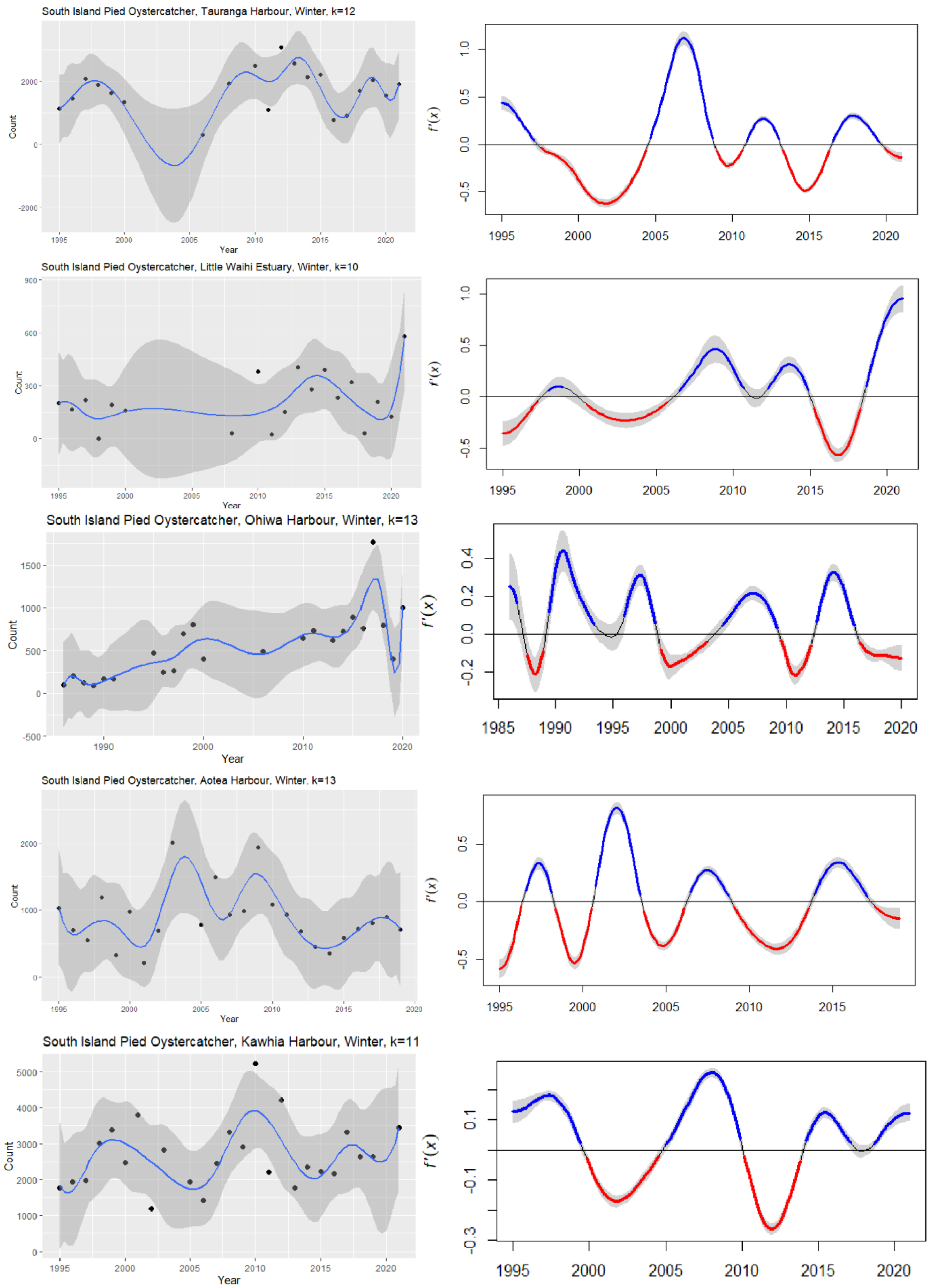


Figure 24. Site-specific trends of South Island Pied Oystercatchers in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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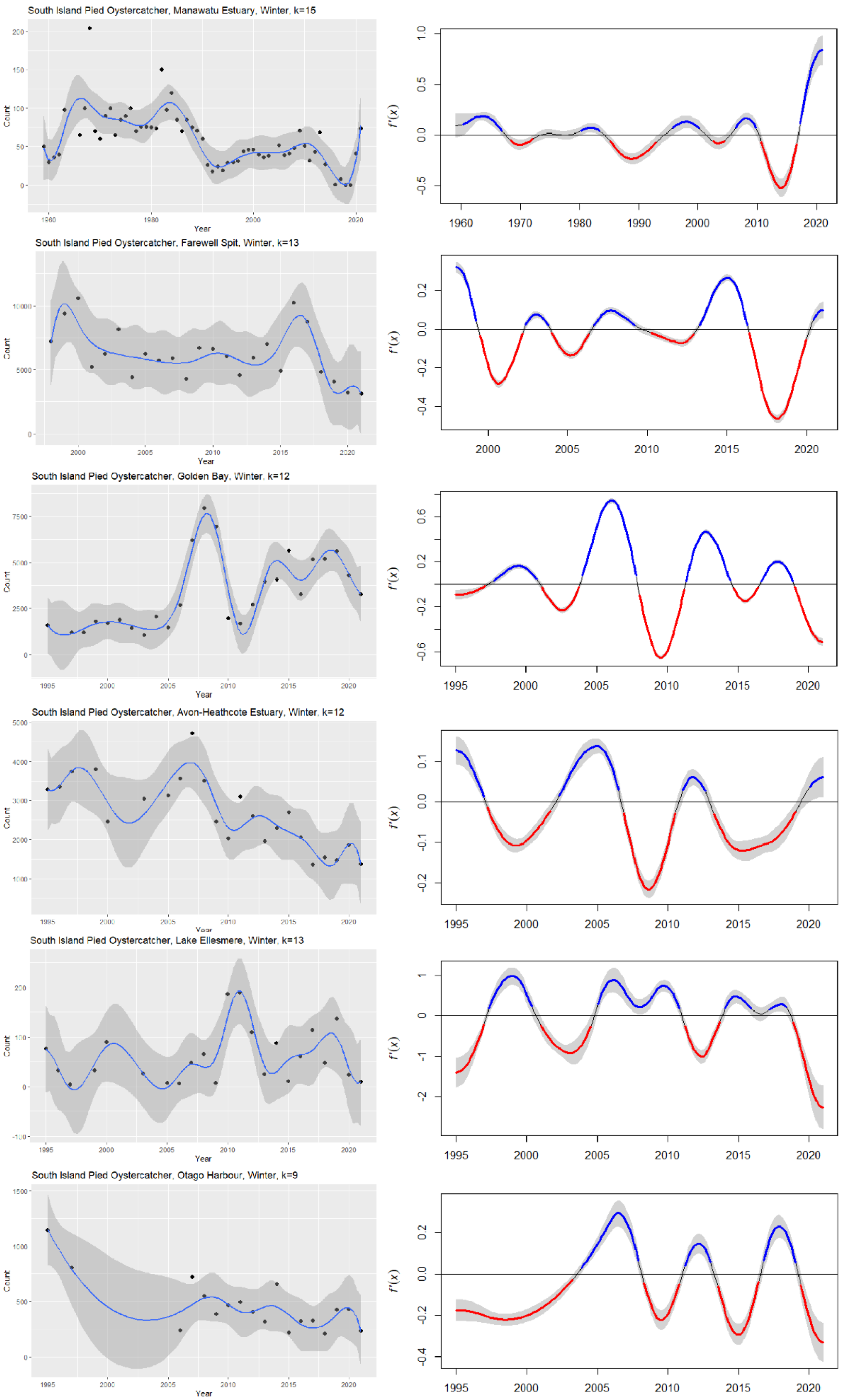

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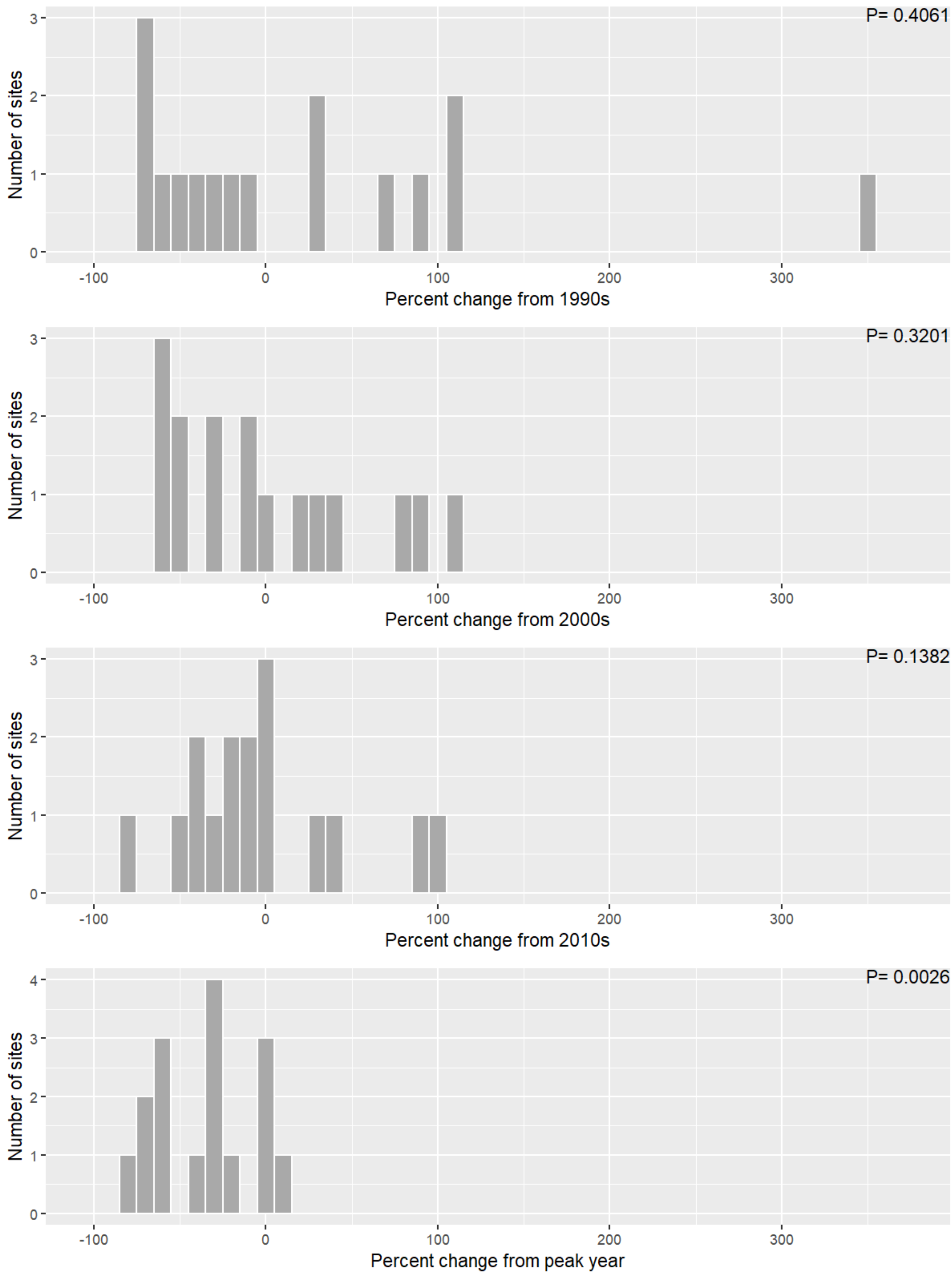


Figure 25. Individual site trends for South Island Pied Oystercatchers in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 19. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for South Island Pied Oystercatchers in winter. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.25	Intercept	437.690	366.136	1.195	0.252
	$F_{1,14}$=1.283, P=0.2764, R^2=0.01849		Latitude	-10.556	9.321	-1.133	0.276
2000s	MaxCount+Lat	0.30	Intercept	375.838355	181.001981	2.076	0.0582
	$F_{2,13}$=3.005, P=0.08459, R^2=0.2109		MaxCount	-0.002254	0.001127	-2.001	0.0667
			Latitude	-9.090235	4.521581	-2.010	0.0656
2010s	Lat+Log(Area.ha)	0.54	Intercept	602.073	114.259	5.269	0.000152
	$F_{2,13}$=14.25, P=0.0005289, R^2=0.6386		Latitude	-12.419	2.581	-4.813	0.000339
			Log(Area. ha)	-15.235	4.742	-3.213	0.006797
MaxYr	MaxCount+Log(Area.ha)	0.64	Intercept	-4.440e+01	4.341e+01	-1.023	0.325
	$F_{2,13}$=0.07317, P=0.9298, R^2=-0.141		MaxCount	-3.143e-04	8.237e-04	-0.382	0.709
			Log(Area. ha)	1.571e+00	6.075e+00	0.259	0.800

Summer

Modelled population trends for SIPO in Summer varied from -92% to an increase of 1674% depending on the site and time period (Table 20, Figure 26). The median trend was significantly different to zero only (being lower) for the year of maximum count (median trends of -4.4% to -36%; significance values are given in Figure 27).

There was an increase in numbers around 2005–2010 at a range of sites, probably indicating good breeding success around that time. The long-term data from the Manukau Harbour suggest there were similar periods about every 10 years, yet at the Firth of Thames there was a slow increase from the 1960s to the 1990s, then a spike around 1995. The Firth of Thames counts matched only one of the four peaks shown at the Manukau Harbour.

Linear models testing trends against population size, habitat area and latitude, found significant or almost-significant relationships for most time periods (Table 21). Latitude was positively related to trend in three models (meaning trends were higher further south in New Zealand), area was positively related to trend in two models (larger sites had better trends) and population size was significant in the trend from the first count year (larger populations tended to have higher trends).

Table 20. South Island Pied Oystercatcher summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	9	602	59	152.6	-50 (-87, 81)	-21 (-78, 165)	21 (-62, 254)	-36 (-83, 113)	-49 (-87, 83)
Mangawhai Estuary	24	6	86	8	13.9	-92 (-98, -69)	-92 (-98, -67)	-33 (-84, 223)	-89 (-97, -57)	-92 (-98, -67)
Kaipara Harbour	27	15	6981	2893	3237.9	94 (-0.74, 286)	96 (17, 231)	137 (40, 288)	12 (-33, 84)	106 (13, 274)
Manukau Harbour	62	15	6375	3663	3508.7	-17 (-47, 34)	-29 (-55, 12)	-37 (-60, -1.5)	-38 (-60, -1.9)	507 (154, 1421)
Firth of Thames	62	15	9169	1464.5	1616.6	-8.3 (-44, 61)	-41 (-67, 2.2)	48 (-21, 161)	-56 (-74, -25)	1674 (242, 7233)
Tauranga Harbour	22	13	992	440.5	444.5	257 (5.5, 1044)	59 (-25, 237)	45 (-21, 184)	0 (0, 0)	234 (6.4, 890)
Raglan Harbour	16	6	271	106.5	115.1		32 (-62, 320)	-23 (-73, 95)	-53 (-82, 17)	22 (-63, 283)
Ohiwa Harbour	26	12	334	134.5	135.7	248 (13, 971)	-49 (-76, 18)	-62 (-81, -24)	-70 (-85, -37)	1368 (153, 8112)
Aotea Harbour	25	14	604	158	203.2	-58 (-83, 4.5)	-72 (-88, -34)	-72 (-88, -34)	-75 (-90, -41)	-59 (-83, 1.3)
Kawhia Harbour	25	10	1033	686	652.7	-26 (-56, 34)	-4.4 (-42, 65)	-15 (-47, 38)	-24 (-53, 22)	-26 (-55, 31)
Farewell Spit	28	12	2797	1417.5	1477.8	-37 (-60, 3.2)	-14 (-44, 31)	-5.5 (-40, 50)	-33 (-57, 5.7)	-37 (-60, 3.6)
Golden Bay	26	14	3548	1522	1484	715 (109, 2972)	82 (3.9, 218)	-9.4 (-46, 50)	-34 (-60, 5.3)	671 (113, 2642)
Avon-Heathcote Estuary	23	4	2243	889	901	25 (-42, 173)	-0.54 (-38, 61)	22 (-29, 116)	-2.4 (-39, 59)	23 (-42, 165)
Lake Ellesmere	24	3	98	18.5	25.9		211 (20, 702)	337 (89, 893)	20 (7.2, 35)	211 (20, 702)
Catlins Lake	17	5	295	58	85.1	-62 (-89, 7.4)	7.3 (-61, 179)	94 (-26, 459)	-60 (-88, 11)	-60 (-88, 11)

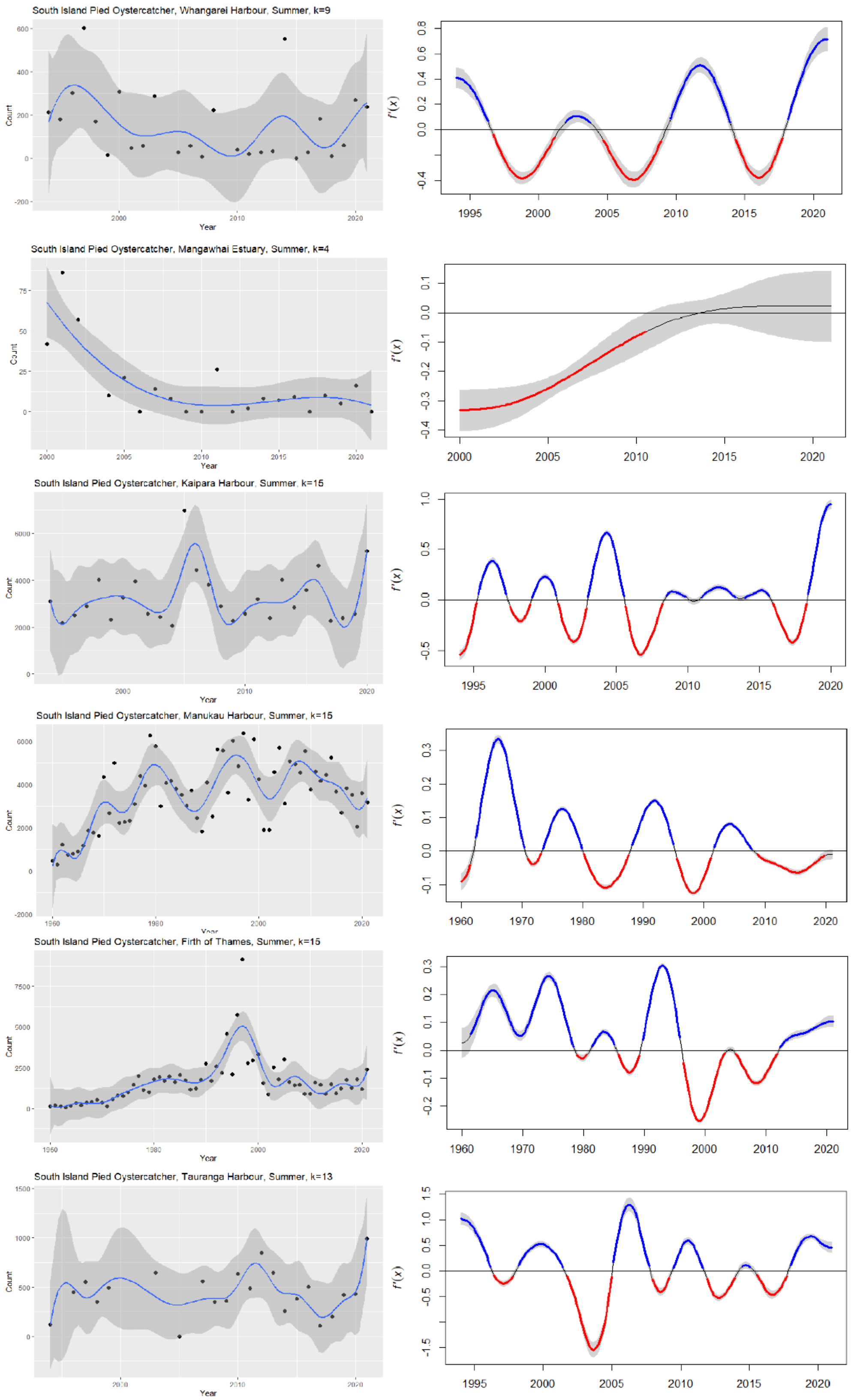
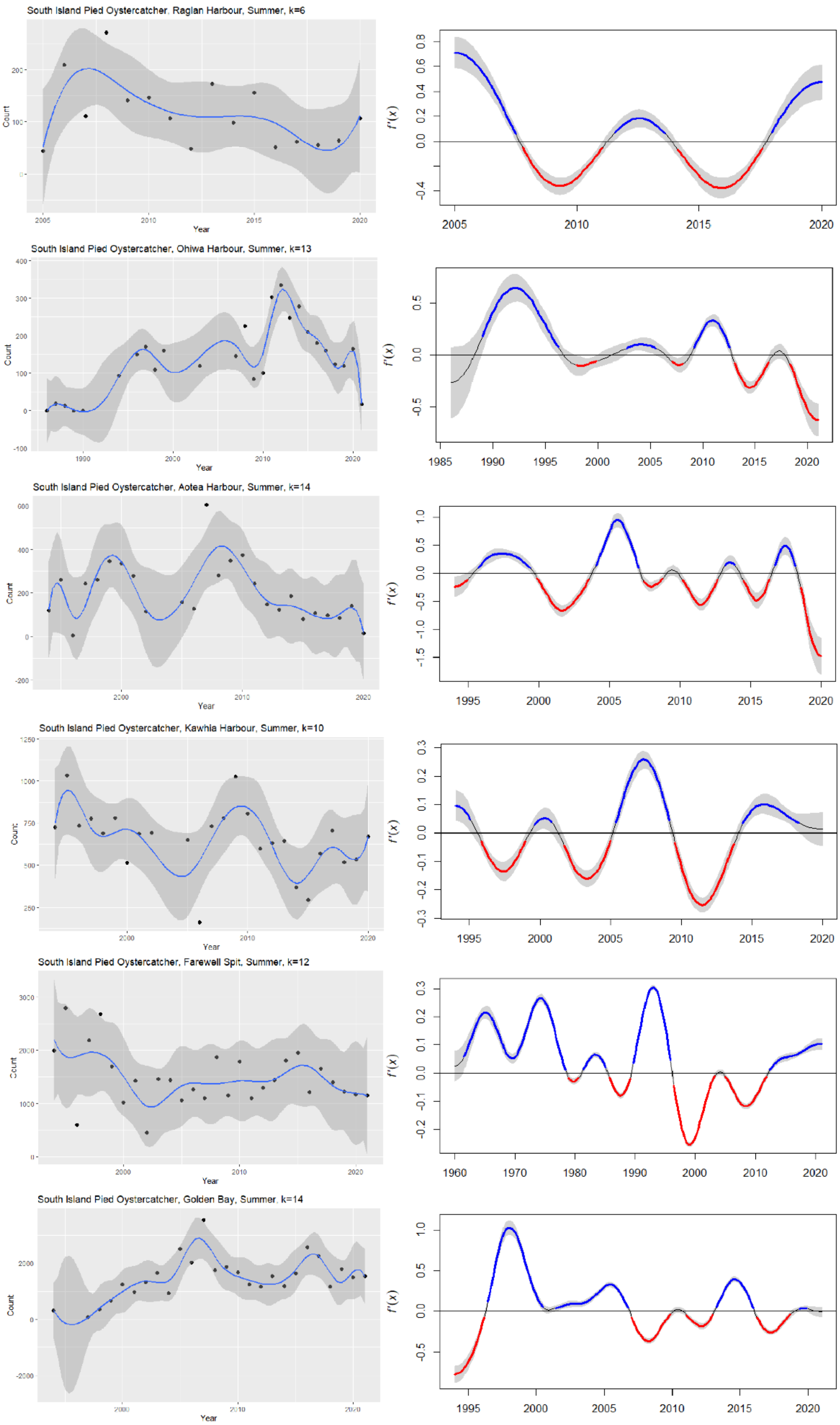


Figure 26. Site-specific trends of South Island Pied Oystercatchers in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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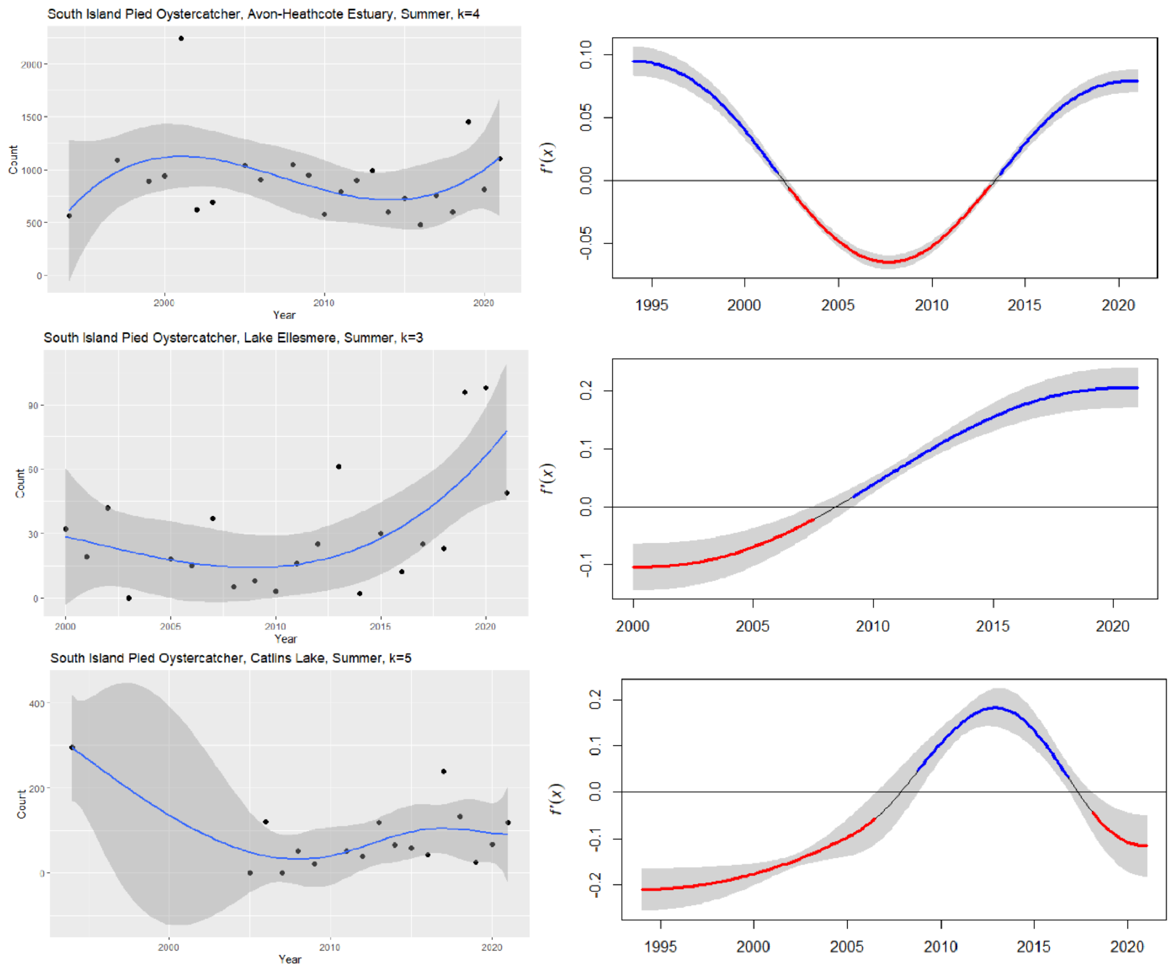

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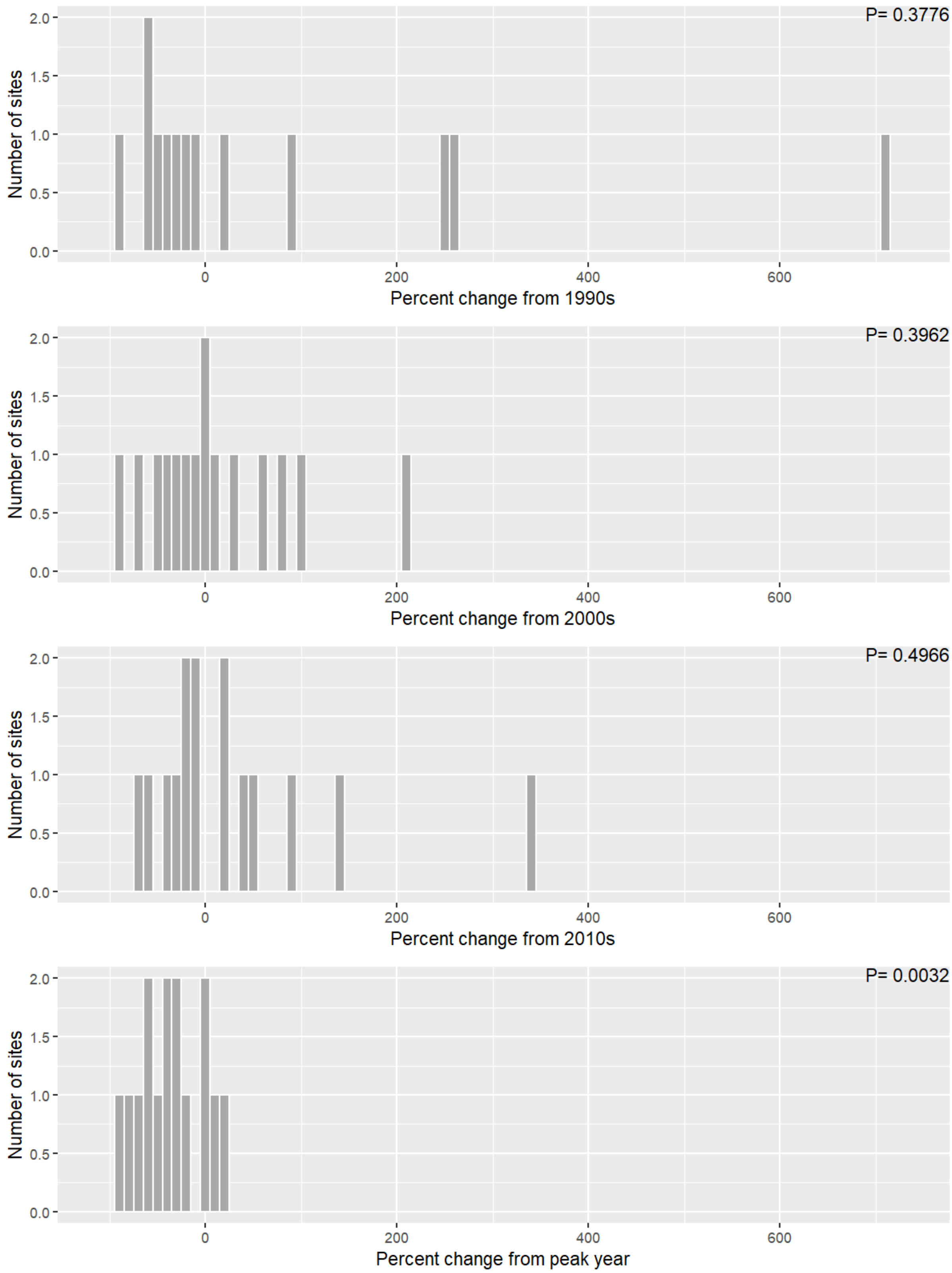


Figure 27. Individual site trends for South Island Pied Oystercatchers in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 21. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for South Island Pied Oystercatchers in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Log(Area.ha)	0.37	Intercept	-229.78	371.54	-0.618	0.549
	$F_{1,11}$=0.6973, P=0.4215, R^2=-0.02588		Log(Area.ha)	38.11	45.64	0.835	0.421
2000s	Lat+Log(Area.ha)	0.47	Intercept	-800.361	282.148	-2.837	0.0150
	$F_{2,12}$=4.251, P=0.04021, R^2=0.3171		Latitude	14.368	5.694	2.523	0.0267
			Log(Area.ha)	31.272	13.699	2.283	0.0415
2010s	Lat	0.53	Intercept	-545.705	307.872	-1.773	0.0997
	$F_{1,13}$=3.516, P=0.08341, R^2=0.1523		Latitude	14.705	7.842	1.875	0.0834
MaxYr	Lat+Log(Area.ha)	0.51	Intercept	-365.107	114.071	-3.201	0.00762
	$F_{2,12}$=5.24, P=0.02313, R^2=0.3772		Latitude	4.999	2.302	2.171	0.05066
			Log(Area.ha)	16.776	5.538	3.029	0.01048

Pied Stilt

Winter

Modelled population trends for Pied Stilt in Winter varied from -97% to an increase of 791% depending on the site and time period (Table 22, Figure 28). The median trend was not significantly different to zero for all periods of the study, except the year of maximum count (median trends of 7.4% to -18%; significance values are given in Figure 29).

Sites showed a range of trend patterns. In the north, Whangarei Harbour and Waitemata Harbour declined in the 2000s then increased again, while the Kaipara Harbour had variable counts and a weak declining trend over the same period. The only other site with a comparable population to the Kaipara Harbour, the Firth of Thames, peaked in the late 1990s at over 6000 birds but then dropped to roughly 3000 birds and subsequently to just over 2000 birds. Sites in the Bay of Plenty tended to increase, while those in the Waikato were stable and those in the lower North Island declined. In the South Island, the large population at Lake Ellesmere remained fairly stable, apart from a large peak in 2011. Given these variable trends, it is unsurprising that linear models found no significant relationships between trends and site factors (Table 23).

Table 22. Pied Stilt winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	12	1096	268	322.1	-21 (-72, 156)	-39 (-76, 41)	67 (-41, 376)	-44 (-79, 32)	-19 (-70, 133)
Mangawhai Estuary	23	13	65	28	25.8	570 (119, 1955)	149 (33, 366)	639 (186, 1797)	0 (0, 0)	454 (107, 1401)
Kaipara Harbour	27	3	6308	3246	3532.5	-17 (-46, 32)	-18 (-42, 22)	-15 (-36, 15)	-18 (-44, 28)	-17 (-46, 32)
Waitemata Harbour	21	14	446	128	146.3	-30 (-67, 44)	28 (-38, 153)	352 (91, 874)	-1.8 (-51, 91)	-25 (-65, 53)
Whitford	15	11	141	85	88.2		217 (8.4, 744)	46 (-25, 169)	0 (0, 0)	217 (13, 716)
Firth of Thames	28	15	6049	2968.5	3230.4	37 (-9.1, 104)	-9.1 (-32, 24)	-37 (-53, -13)	-53 (-65, -39)	15 (-22, 63)
Tauranga Harbour	20	14	1549	645	742.8	250 (33, 816)	203 (36, 562)	51 (-19, 166)	10 (-37, 90)	247 (39, 762)
Little Waihi Estuary	19	11	776	213	250.3	103 (-18, 487)	631 (143, 2058)	791 (207, 2472)	0 (0, 0)	114 (-6.5, 475)
Ohiwa Harbour	24	13	418	118	157.5	324 (148, 622)	380 (156, 824)	22 (-27, 97)	-21 (-50, 21)	90 (1, 234)
Aotea Harbour	24	12	700	47.5	85	-80 (-99, 486)	-84 (-99, 254)	-84 (-99, 242)	-97 (-100, -52)	-81 (-99, 359)
Kawhia Harbour	25	13	580	189	224.1	-31 (-77, 97)	-26 (-69, 72)	-14 (-46, 33)	-6.8 (-25, 14)	-31 (-77, 96)
Ahuriri Estuary	26	11	1632	605	660.1	-67 (-84, -33)	-29 (-64, 40)	-18 (-60, 64)	-66 (-83, -30)	-66 (-83, -30)
Manawatū Estuary	58	14	450	188	188.2	-69 (-85, -37)	-50 (-76, 4.2)	-34 (-67, 21)	-72 (-85, -41)	-62 (-83, -18)
Farewell Spit	26	12	66	14.5	21.9	-80 (-94, -33)	-73 (-90, -28)	-50 (-70, -16)	-76 (-92, -30)	-79 (-93, -32)
Golden Bay	26	13	104	52	53	-8.5 (-51, 74)	-12 (-44, 43)	54 (-7.4, 180)	-12 (-44, 43)	-9 (-50, 70)
Lake Ellesmere	23	14	2572	623	755.4	18 (-52, 181)	15 (-44, 129)	7.4 (-26, 54)	6.7 (-24, 48)	18 (-52, 179)
Otago Harbour	18	11	115	28	38.1	-14 (-93, 777)	8.9 (-95, 1819)	-43 (-89, 251)	-55 (-89, 82)	-13 (-92, 742)

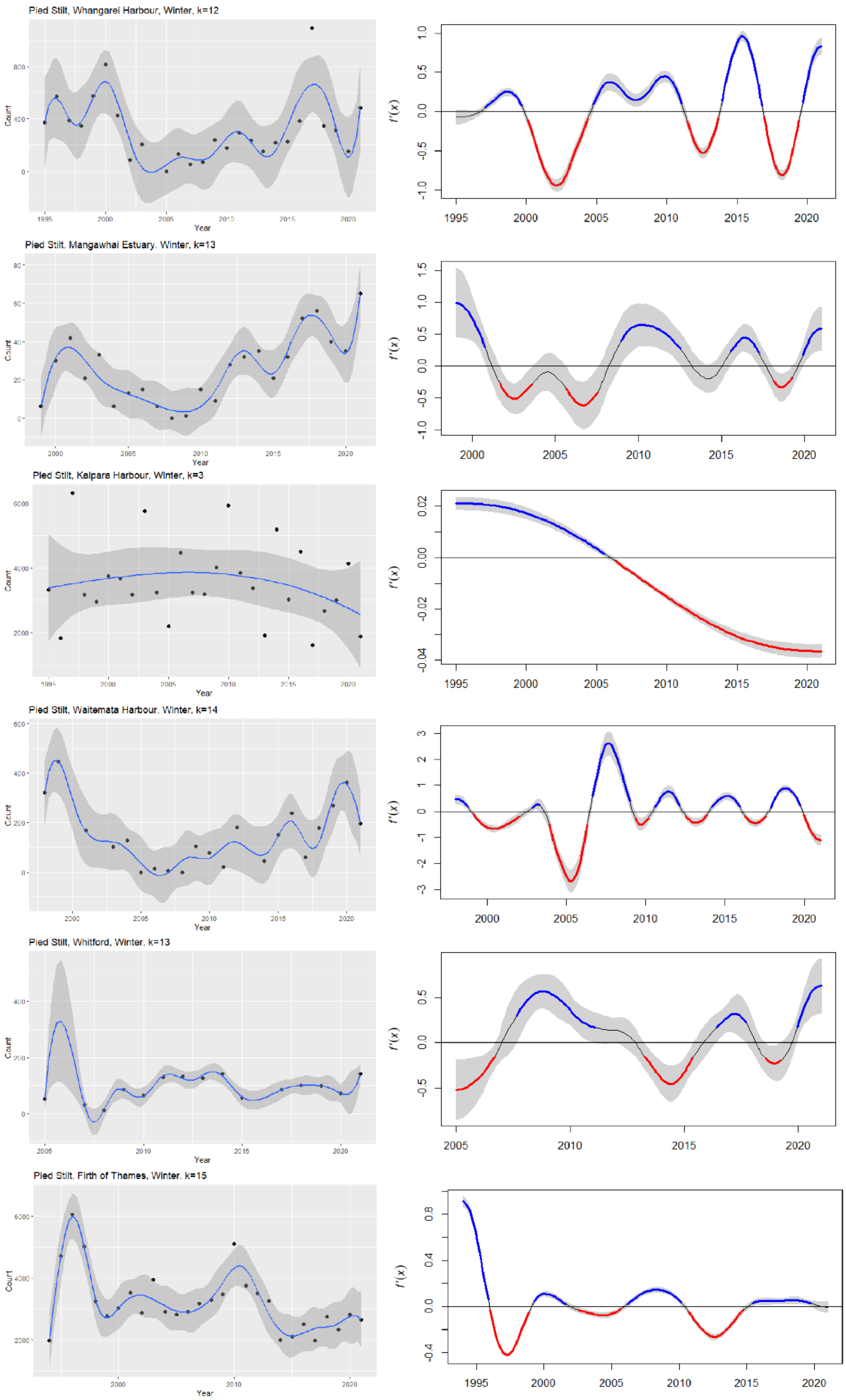
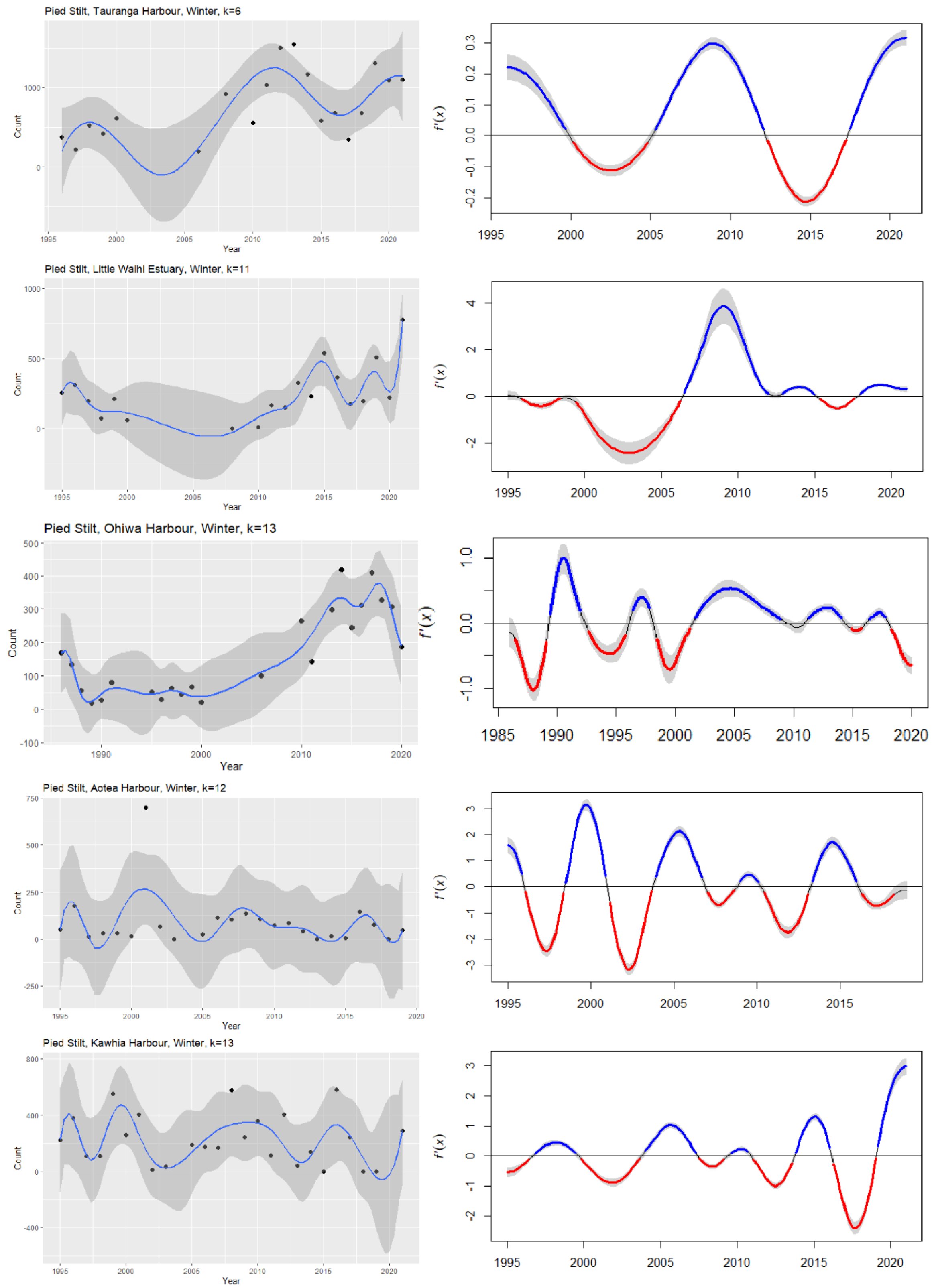


Figure 28. Site-specific trends of Pied Stilts in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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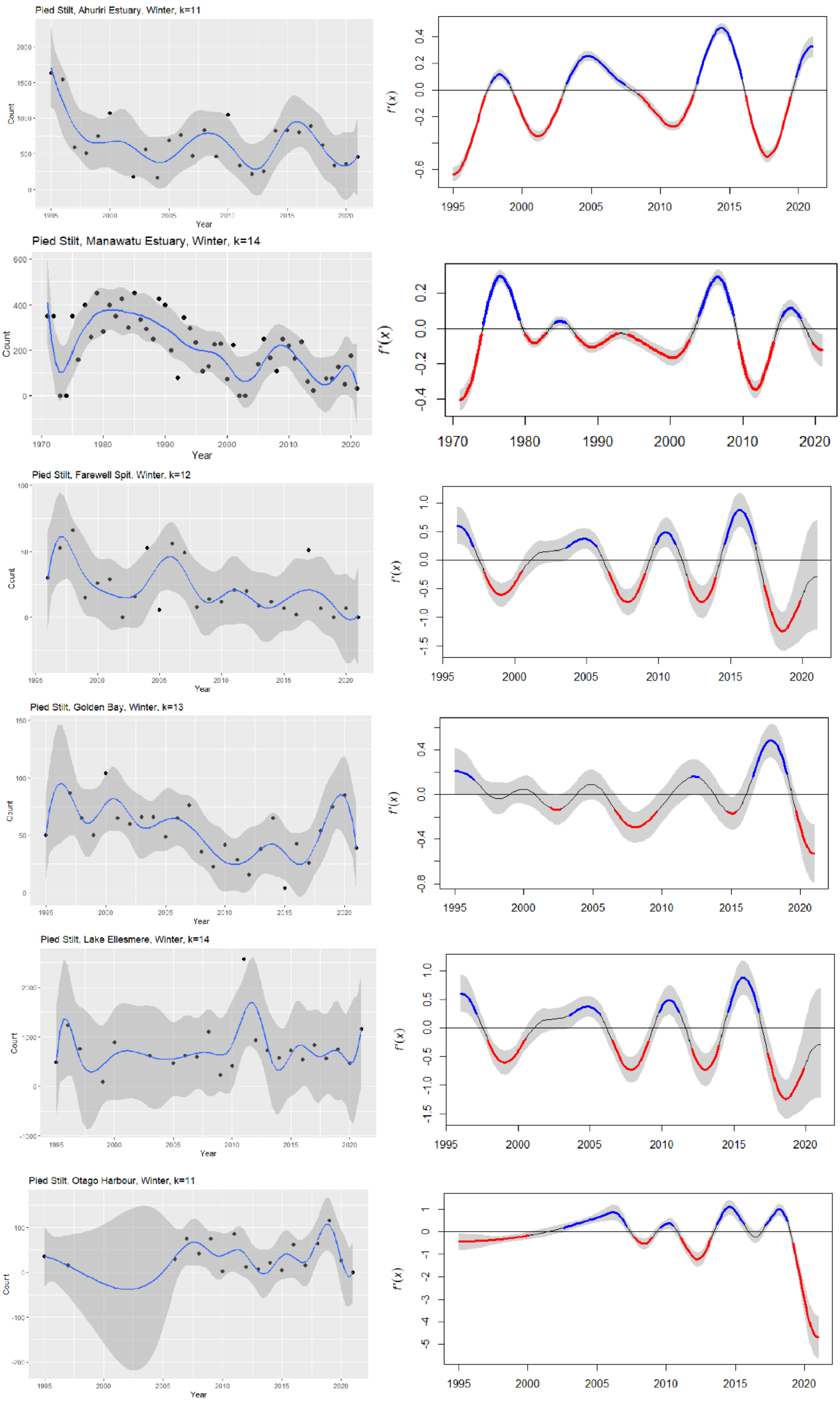

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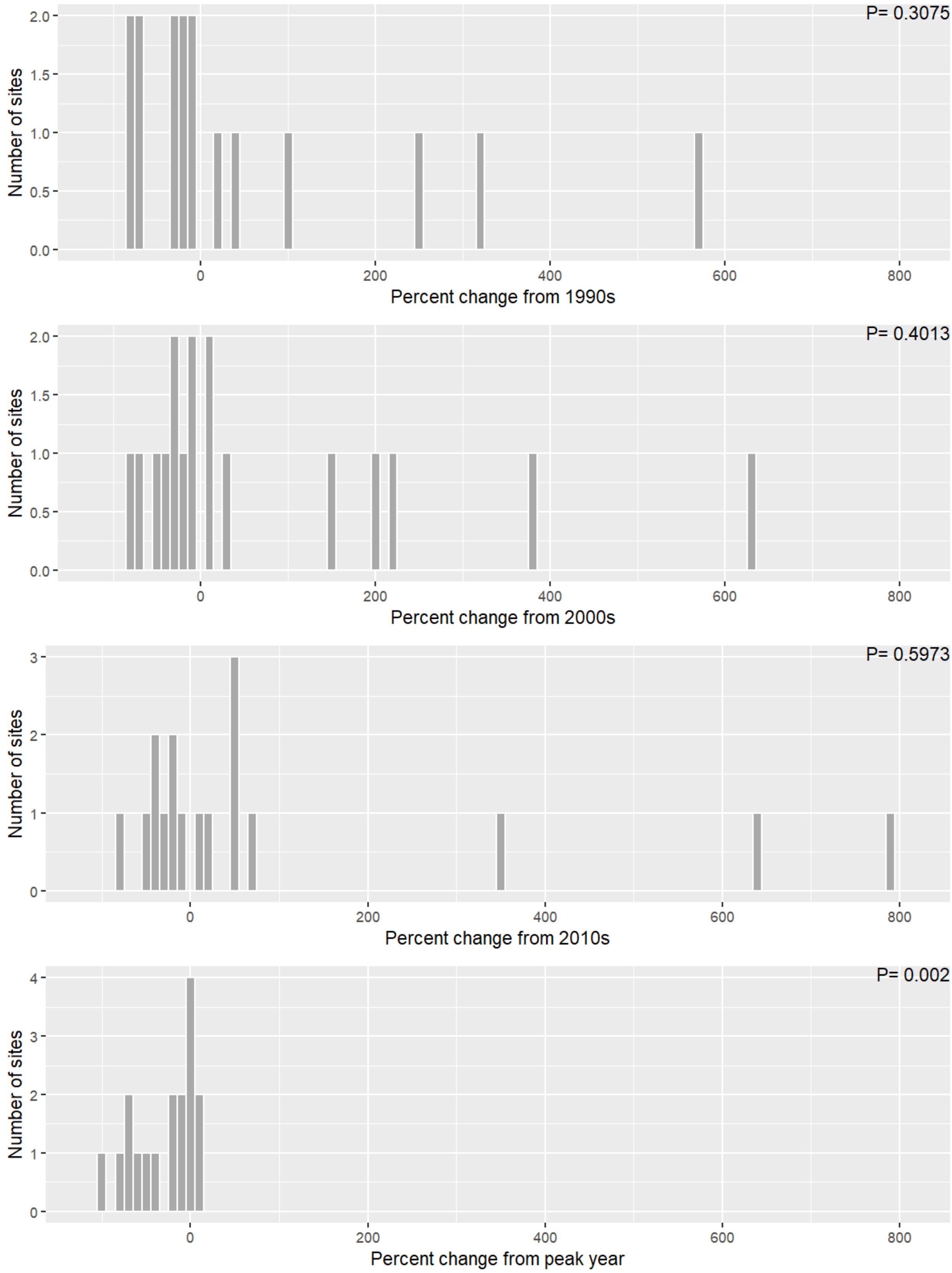


Figure 29. Individual site trends for Pied Stilts in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 23. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Pied Stilts in winter. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.41	Intercept	872.21	633.80	1.376	0.190
	$F_{1,14}$=1.669, P=0.2173, R^2=0.04271		Latitude	-21.00	16.25	-1.292	0.217
2000s	Lat	0.29	Intercept	695.12	676.11	1.028	0.320
	$F_{1,15}$=0.8425, P=0.3732, R^2=-0.009942		Latitude	-15.96	17.39	-0.918	0.373
2010s	Log(Area.ha)	0.25	Intercept	467.52	263.59	1.774	0.0964
	$F_{1,15}$=2.024, P=0.1753, R^2=0.06016		Log(Area.ha)	-49.86	35.05	-1.423	0.1753
MaxYr	Lat	0.31	Intercept	75.361	121.838	0.619	0.546
	$F_{1,15}$=0.7479, P=0.4008, R^2=-0.01601		Latitude	-2.710	3.134	-0.865	0.401

Summer

Modelled population trends for Pied Stilts in Summer varied from -95% to an increase of 138% depending on the site and time period (Table 24, Figure 30). The median trend was not significantly different to zero for all periods of the study, except the year of maximum count (median trends of 24% to -38%; significance values are given in Figure 31).

Both of the longer-term sites (Manukau Harbour and the Firth of Thames) showed a stable population from the start of counts until the 1980s and have since been experiencing a gradual decline (though the Firth of Thames has shown some resurgence in numbers in recent years). Other sites with large stilt populations in summer have also tended to decline (Whangarei Harbour Kaipara Harbour, Ahuriri Estuary).

Linear models found no significant relationships between trends and population size, habitat area and latitude (Table 25).

Table 24. Pied Stilt summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	14	598	74.5	155.7	-95 (-99, -78)	-92 (-98, -60)	-44 (-89, 210)	-95 (-99, -78)	-95 (-99, -78)
Waipu	24	10	30	4.5	7.6	-32 (-93, 558)	-68 (-94, 75)	-28 (-88, 418)	-67 (-94, 86)	-35 (-92, 510)
Mangawhai Estuary	24	8	70	19	22.8	140 (-24, 667)	96 (-19, 381)	42 (-10, 128)	21 (-5.8, 57)	138 (-24, 652)
Kaipara Harbour	27	12	652	267	299.1	29 (-59, 288)	3.4 (-55, 153)	41 (-40, 269)	-3.7 (-62, 123)	24 (-59, 257)
Manukau Harbour	62	15	1767	683	748.2	-64 (-77, -44)	-51 (-68, -30)	-31 (-47, -11)	-76 (-84, -63)	-76 (-84, -63)
Firth of Thames	62	15	2603	740	815.5	5.6 (-43, 99)	25 (-35, 142)	16 (-32, 97)	-20 (-55, 40)	2.8 (-53, 109)
Little Waihi Estuary	19	9	455	70	118.7	-23 (-92, 633)	24 (-81, 888)	-34 (-88, 218)	-88 (-98, -47)	-26 (-91, 468)
Lake Wairarapa	12	6	760	368.5	388.6			60 (-19, 225)	4.8 (-2.1, 12)	59 (-19, 221)
Little Waihi Estuary	19	9	455	70	118.7	-23 (-92, 633)	24 (-81, 888)	-34 (-88, 218)	-88 (-98, -47)	-26 (-91, 468)
Ohiwa Harbour	26	6	82	35	34.7	67 (-25, 262)	45 (-17, 148)	22 (-14, 71)	1.9 (-1.6, 5.8)	75 (-28, 327)
Kawhia Harbour	25	10	123	50	59.4	23 (-54, 232)	38 (-35, 236)	45 (-24, 177)	45 (-24, 177)	24 (-53, 227)
Ahuriri Estuary	25	12	847	336	358.1	-76 (-90, -51)	-63 (-83, -20)	-62 (-82, -23)	-64 (-83, -30)	-76 (-90, -50)
Farewell Spit	28	10	32	3.5	6.4	-87 (-98, 150)	-80 (-98, 176)	-78 (-98, 206)	-91 (-99, 20)	-86 (-98, 159)
Avon-Heathcote Estuary	23	12	191	54	61	97 (-66, 1070)	73 (-48, 411)	-56 (-85, 8.7)	-56 (-85, 8.7)	95 (-63, 978)

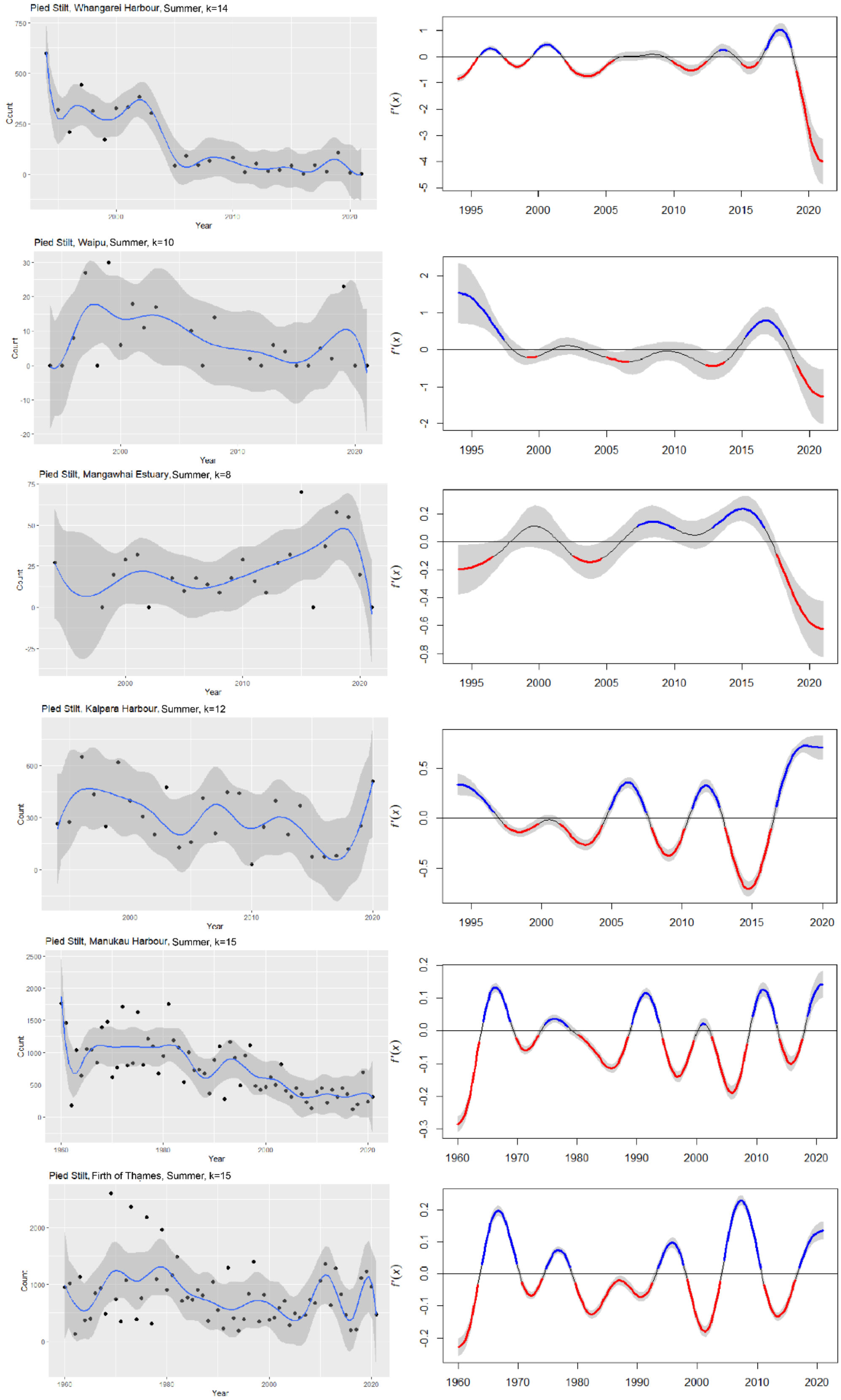
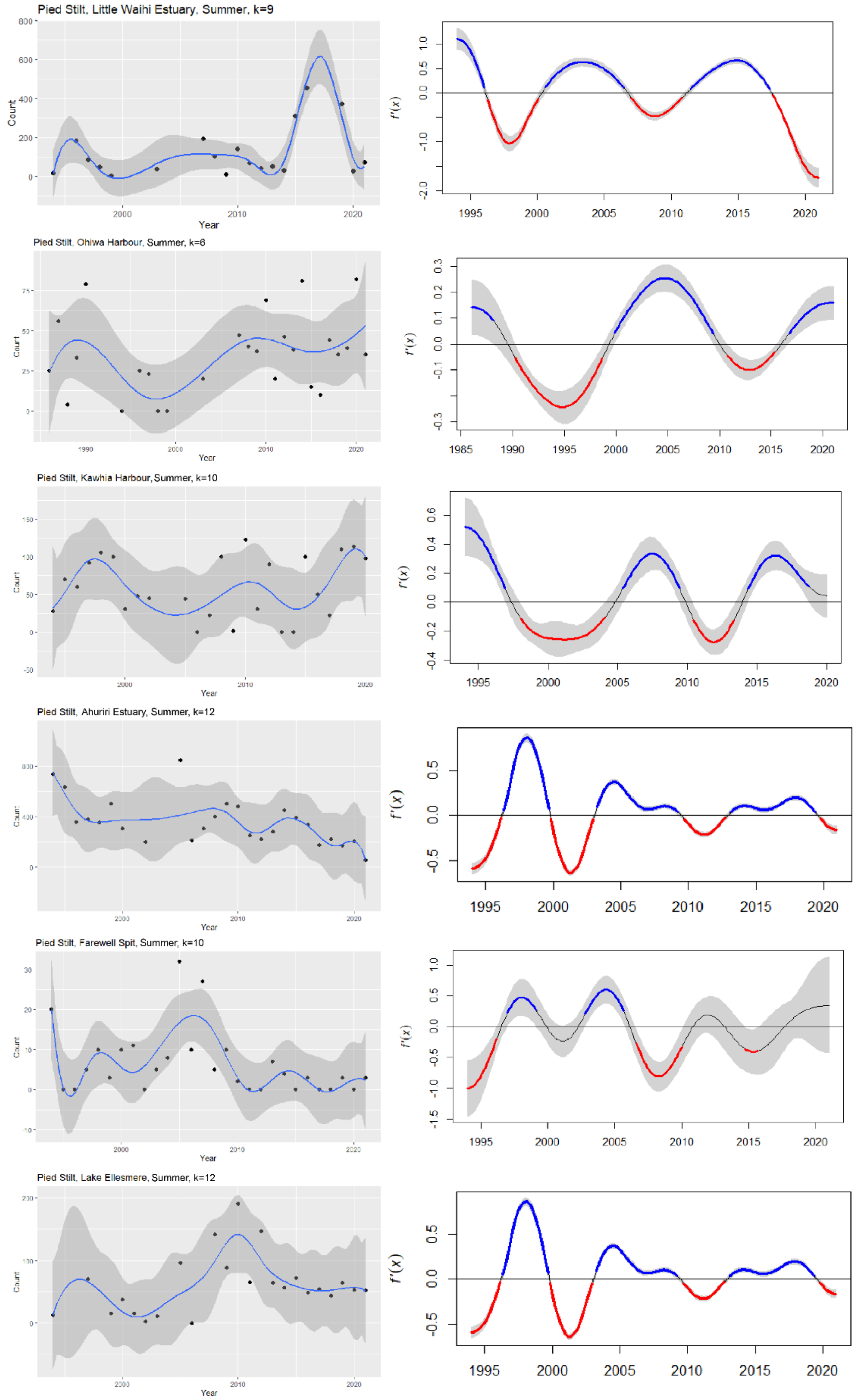


Figure 30. Site-specific trends of Pied Stilts in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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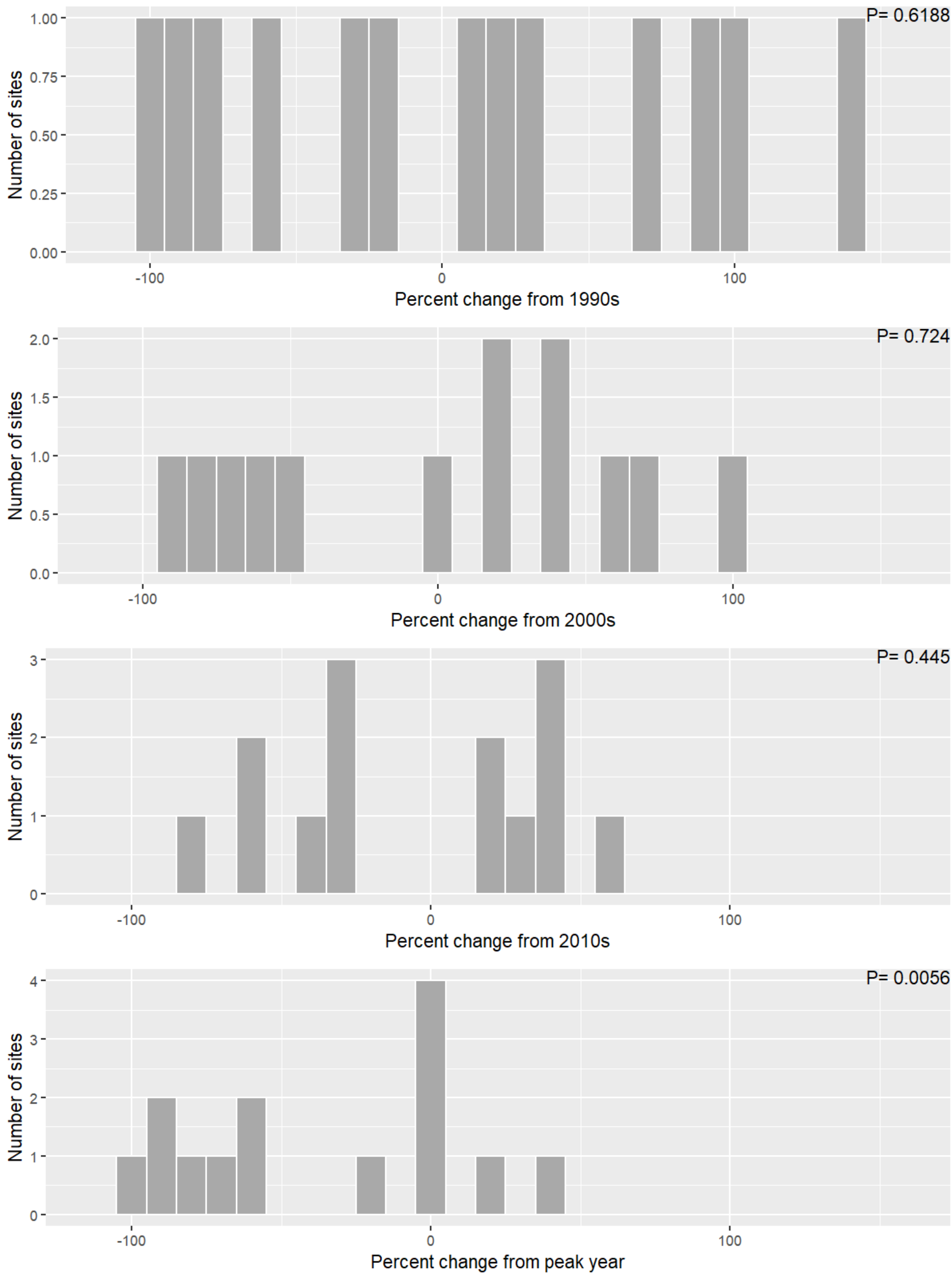


Figure 31. Individual site trends for Pied Stilts in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 25. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Pied Stilts in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.40	Intercept	-313.761	316.737	-0.991	0.343
	$F_{1,11}$=1.023, P=0.3336, R^2=0.001882		Latitude	8.313	8.221	1.011	0.334
2000s	Lat	0.42	Intercept	-288.422	262.739	-1.098	0.296
	$F_{1,11}$=1.22, P=0.2928, R^2=0.01804		Latitude	7.534	6.819	1.105	0.293
2010s	MaxCount	0.32	Intercept	-12.872767	16.104575	-0.799	0.440
	$F_{1,12}$=0.5495, P=0.4728, R^2=-0.0359		MaxCount	0.008522	0.011496	0.741	0.473
MaxYr	MaxCount	0.30	Intercept	-39.212861	16.503222	-2.376	0.035
	$F_{1,12}$=0.2132, P=0.6526, R^2=-0.06442		MaxCount	0.005439	0.011781	0.462	0.653

Banded Dotterel

Winter

Modelled population trends for Banded Dotterels in winter varied from -100% to an increase of 963% depending on the site and time period (

Table 26, Figure 32). The median trend was not significantly different to zero for all periods of the study, except the year of maximum count (median trends of 0.92% to -10.85%; significance values are given in Figure 33).

Of the sites with large populations (in descending order), Lake Ellesmere showed a slow decline from >1500 to 500 birds, Farewell Spit and Kaipara Harbour have been essentially stable, while the Manukau Harbour increased in the 1960s and 1970s but declined slowly to the 2000s before alternating between short-term increases and decreases. Some changes in site populations could be due to local shifts, such as from Westhaven Inlet and Golden Bay to Farewell Spit in the 2000s.

There were no significant associations between local population trend and population size, habitat area and latitude (Table 27).

Table 26. Banded Dotterel winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	10	564	173.5	209.4	-28 (-76, 120)	5.5 (-60, 155)	61 (-38, 264)	48 (-34, 202)	-26 (-75, 119)
Mangawhai Estuary	23	12	182	99	98	76 (-9, 260)	71 (-8.6, 236)	32 (-4.5, 89)	5.2 (-0.85, 12)	75 (-8.9, 256)
Kaipara Harbour	27	13	1087	650	669.9	11 (-24, 69)	8.6 (-20, 52)	4.4 (-11, 25)	4 (-10, 22)	11 (-24, 69)
Waitemata Harbour	21	13	253	23	44.2	138 (-61, 1474)	119 (-57, 1110)	51 (-36, 268)	25 (-22, 103)	136 (-60, 1431)
Whitford	15	9	73	46	43.7		-53 (-71, -26)	-32 (-55, 3.5)	-39 (-62, -4.5)	-39 (-62, -4.5)
Manukau Harbour	61	11	1076	507	535.5	-15 (-43, 28)	-13 (-39, 31)	-6.9 (-30, 21)	-5.6 (-26, 20)	67 (0.37, 188)
Firth of Thames	28	11	354	153	165.2	200 (48, 496)	89 (13, 204)	10 (-29, 71)	0.78 (-14, 18)	194 (49, 471)
Tauranga Harbour	20	13	628	199.5	216.3	-60 (-88, 22)	-52 (-81, 17)	-32 (-59, 8.6)	-37 (-65, 10)	-60 (-88, 22)
Little Waihi Estuary	19	11	148	41	58.7	-29 (-73, 139)	-24 (-65, 101)	-13 (-43, 44)	-7.6 (-26, 22)	-29 (-73, 137)
Ohiwa Harbour	24	12	676	267	259.3	-58 (-80, -16)	-18 (-64, 77)	8.6 (-47, 106)	-64 (-82, -25)	-68 (-86, -29)
Aotea Harbour	24	13	358	99.5	125.4	1098 (19, 13969)	52 (-53, 413)	318 (16, 1695)	50 (-56, 434)	963 (19, 10283)
Kawhia Harbour	25	11	653	405	399.5	0.92 (-43, 74)	-8.7 (-45, 44)	-28 (-56, 10)	-28 (-56, 10)	0.98 (-41, 65)
Manawatū Estuary	58	12	210	87.5	82.6	-92 (-98, -73)	-94 (-98, -79)	-91 (-97, -70)	-95 (-99, -83)	-96 (-99, -87)
Farewell Spit	26	9	1921	769.5	844.7	47 (-29, 199)	14 (-32, 100)	-14 (-47, 45)	-23 (-56, 28)	43 (-28, 176)
Westhaven Inlet	19	12	406	151	148		-53 (-100, 1100000000)	-100 (-100, 20289)	-100 (-100, 19631)	-77 (-100, 3100000)
Golden Bay	26	10	300	70.5	70.8	-79 (-95, -12)	-80 (-95, -31)	-64 (-92, 50)	-82 (-95, 43)	-79 (-95, -19)
Avon-Heathcote Estuary	23	12	125	71	75.6	-37 (-66, 15)	-14 (-45, 48)	-31 (-57, 13)	-40 (-63, -3.7)	-35 (-64, 15)
Lake Ellesmere	23	5	2168	826	912.4	-61 (-79, -30)	-53 (-72, -25)	-33 (-48, -14)	-59 (-78, -29)	-61 (-79, -30)
Otago Harbour	18	9	144	67.5	59.8	34 (-64, 388)	26 (-56, 255)	13 (-35, 94)	16 (-40, 119)	34 (-64, 381)
Catlins Lake	14	5	63	31	32	36 (-74, 584)	28 (-66, 366)	14 (-43, 124)	9.8 (-34, 80)	35 (-74, 571)
Invercargill	17	12	567	51	95.1	21 (-80, 723)	60 (-76, 908)	299 (-27, 2125)	26 (-15, 88)	23 (-79, 730)
Awarua Bay	19	11	332	22	45.4		-32 (-81, 182)	-23 (-68, 103)	-13 (-46, 47)	-31 (-81, 179)

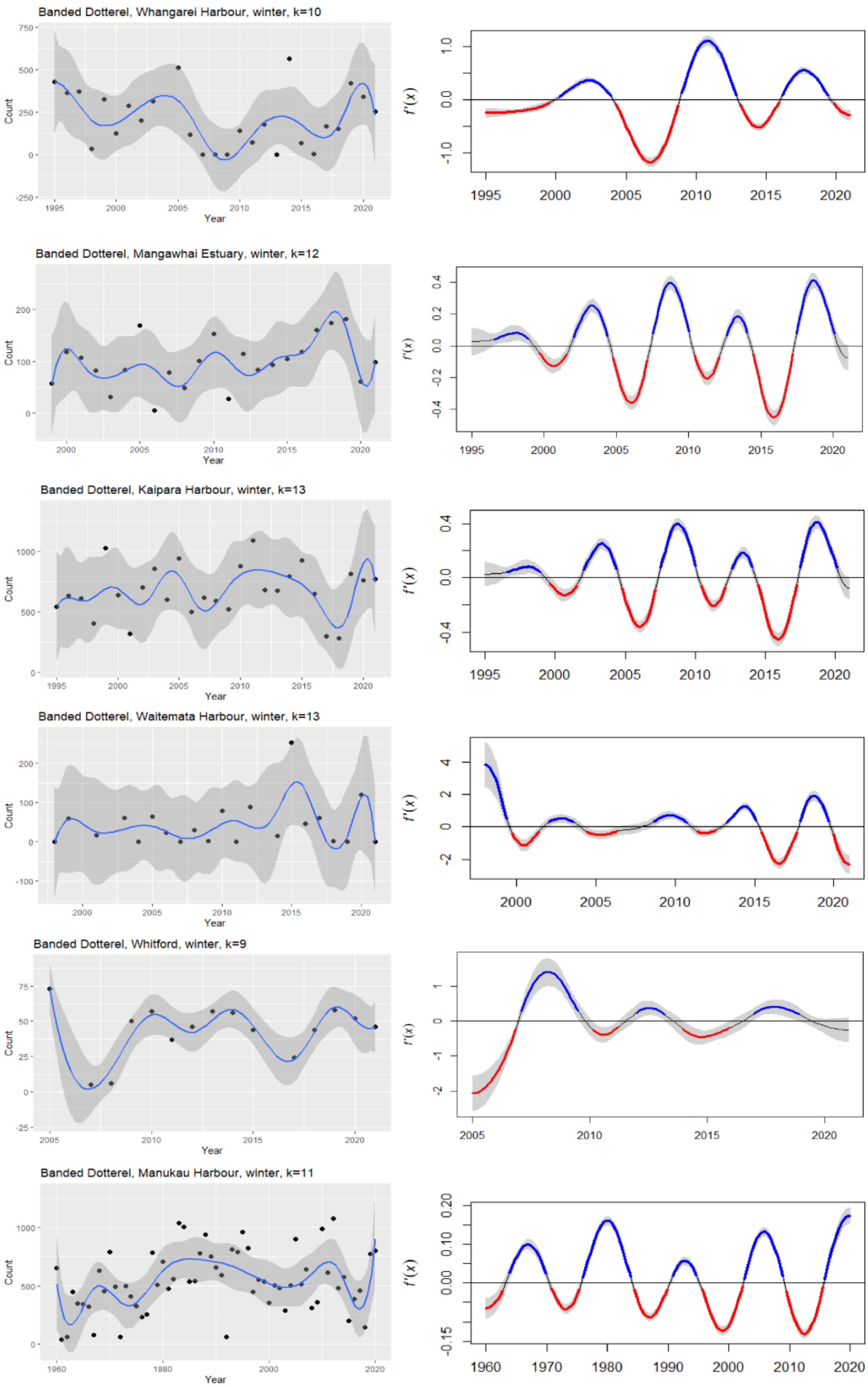
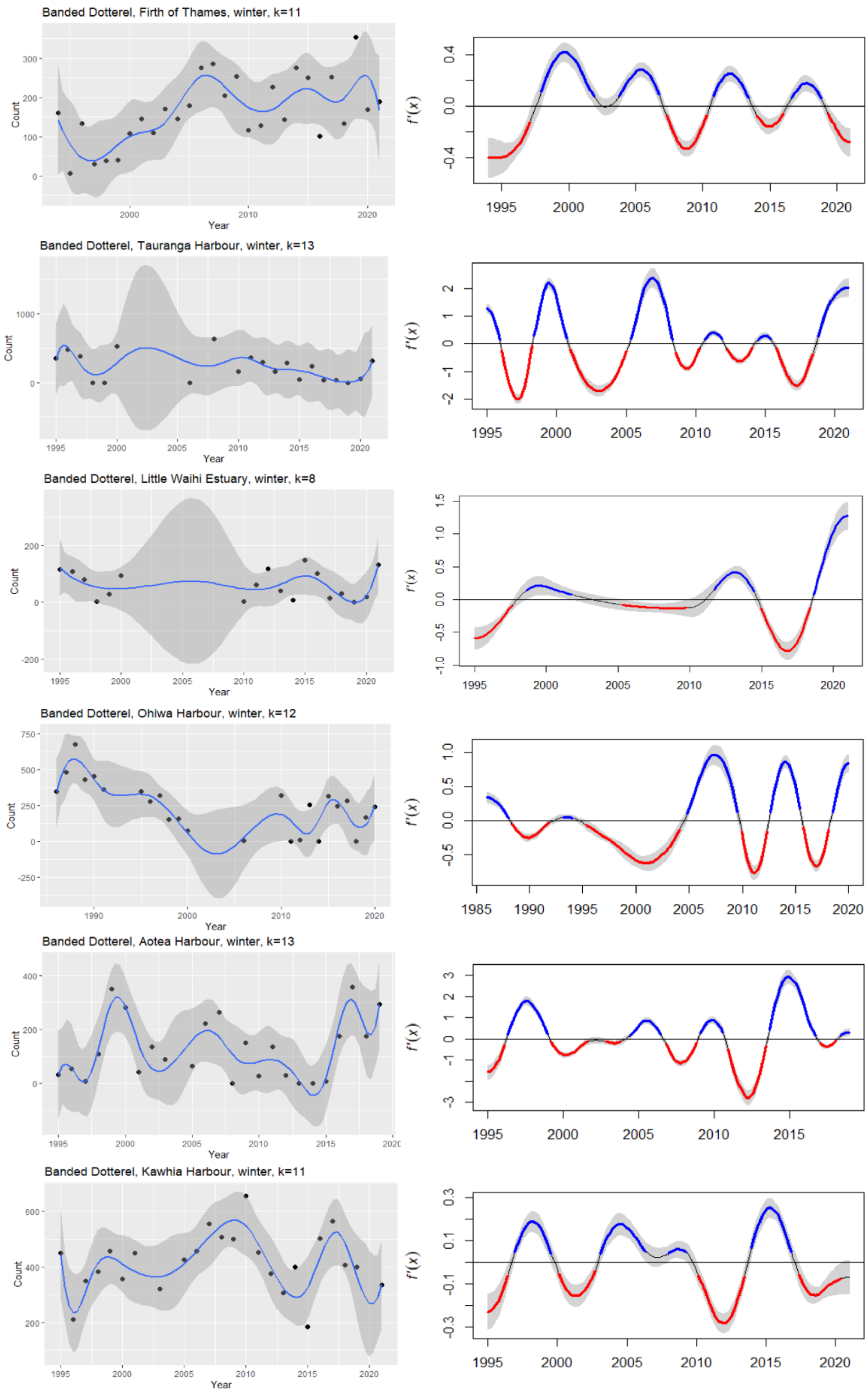


Figure 32. Site-specific trends of Banded Dotterels in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.



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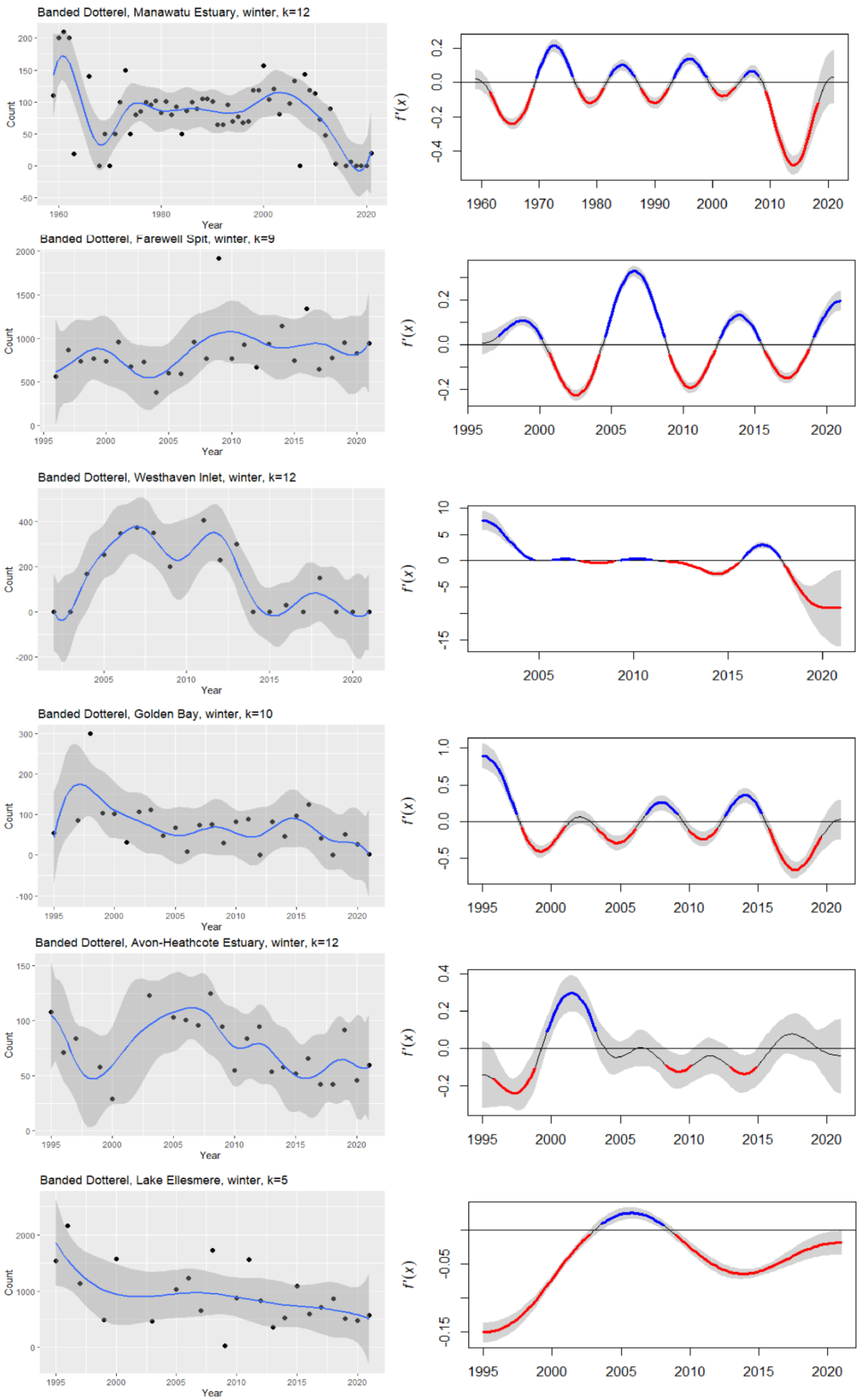

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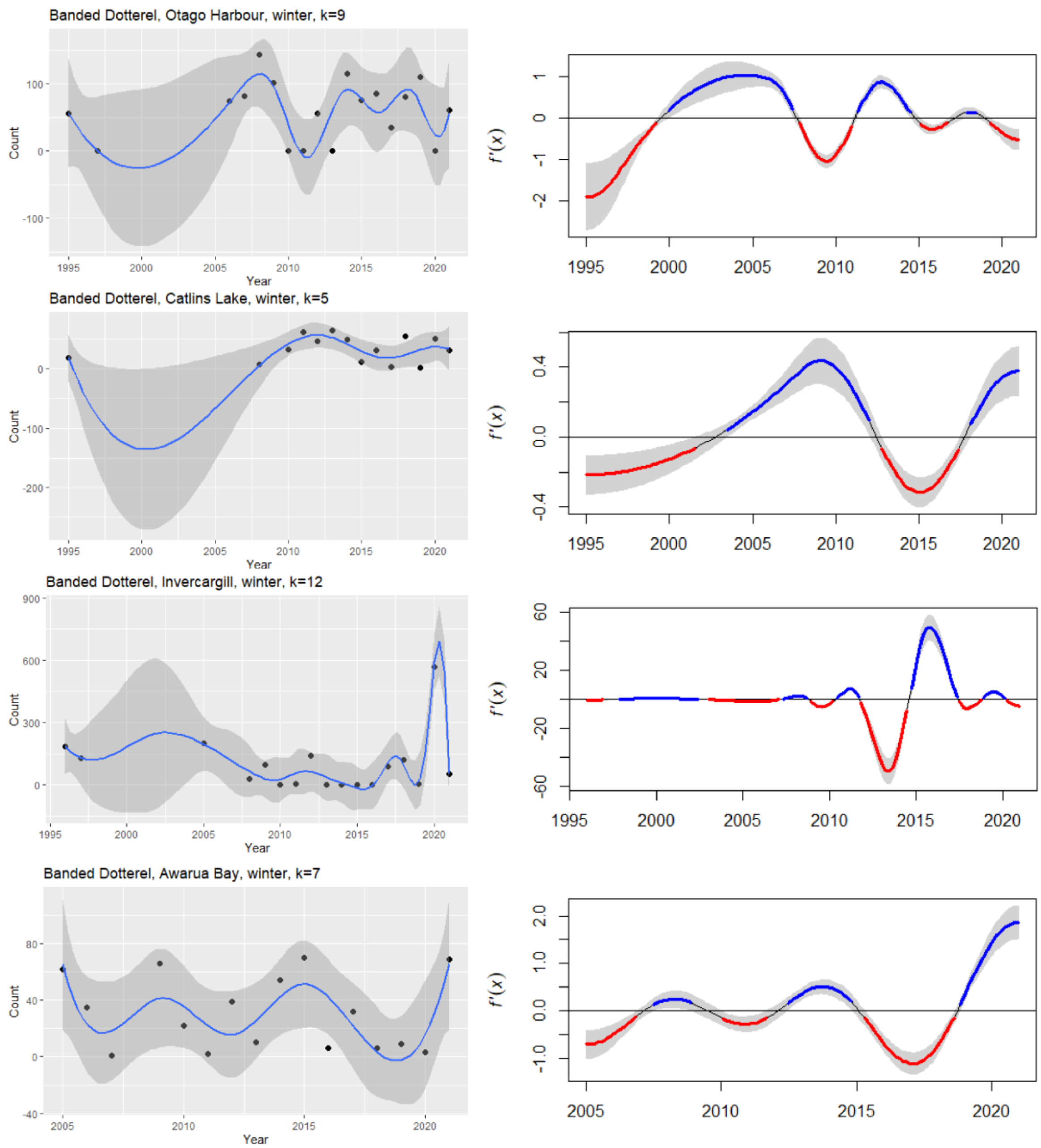

Figure 32 Continued

Table 27. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Banded Dotterels in winter. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.30	Intercept	509.13	685.38	0.743	0.468
	$F_{1,17}$=0.4265, P=0.5225, R^2=-0.03291		Latitude	-11.20	17.14	-0.653	0.522
2000s	Lat	0.27	Intercept	65.574	131.735	0.498	0.624
	$F_{1,20}$=0.2574, P=0.6175, R^2=-0.03666		Latitude	-1.663	3.277	-0.507	0.617
2010s	Lat	0.27	Intercept	-65.930	246.040	-0.268	0.791
	$F_{1,20}$=0.1107, P=0.7428, R^2=-0.04422		Latitude	2.037	6.121	0.333	0.743
MaxYr	Log(Area.ha)	0.28	Intercept	-54.260	50.321	-1.078	0.294
	$F_{1,20}$=0.5206, P=0.4789, R^2=-0.02336		Log(Area. ha)	4.746	6.577	0.722	0.479

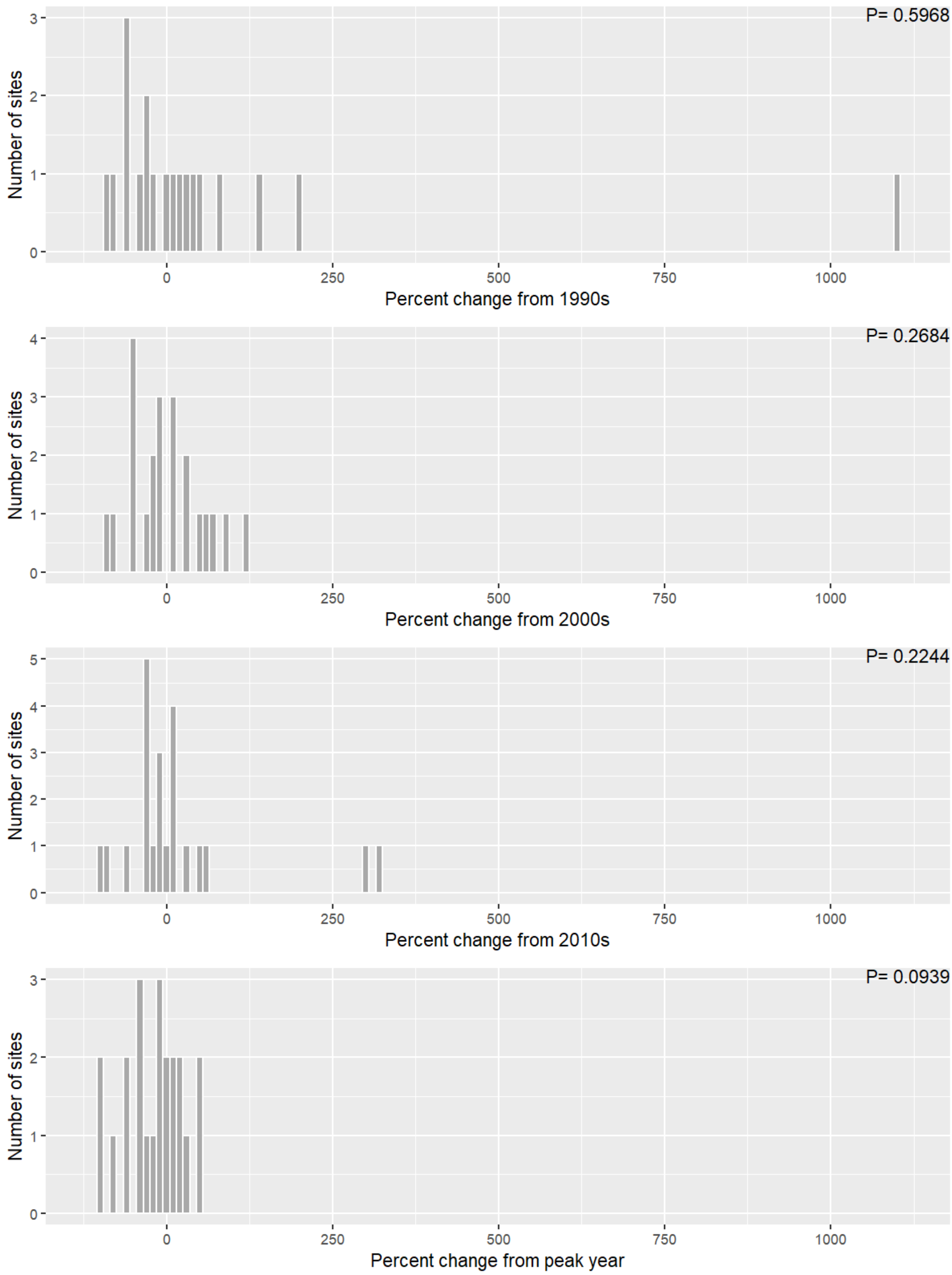

Figure 33. Individual site trends for Banded Dotterels in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Summer

Modelled population trends for Banded Dotterel in summer varied from -89% to an increase of 468% depending on the site and time period (Table 28, Figure 34). The median trend was not significantly different to zero for all periods of the study, except the year of maximum count (median trends of -1.1% to -73%; significance values are given in Figure 35).

Most sites showed an increase in population in the years leading up to 2010 before experiencing a decline. This included Ahuriri Estuary which in the few years preceding 2010 saw their population increase tenfold. Most sites experienced a decline around the start of the 2000s, with Lake Ellesmere experiencing a decline from over 500 birds to less than 200. Longer-term declines are evident in the series from the Manukau Harbour and the Firth of Thames, where the species has virtually been lost as a breeding bird.

Models of trends versus population size, habitat area and latitude included no consistent covariates (Table 29). Three included a negative but non-significant relationship with latitude and one a non-significant relationship with maximum count. The final model (trends from the 2000s onward) had a significant negative relationship with area, suggesting that larger sites experienced proportionally greater declines than did the smaller sites.

Table 28. Banded Dotterel summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Mangawhai Estuary	24	3	12	5	5.3	58 (-45, 354)	24 (-44, 172)	-2.7 (-46, 75)	-1.7 (-14, 12)	57 (-44, 345)
Kaipara Harbour	27	8	332	22	45.4	775 (-49, 26475)	-98 (-100, 618)	125 (-73, 2884)	-73 (-94, 47)	468 (-51, 12095)
Manukau Harbour	62	12	44	2	5.7	-34 (-95, 699)	-78 (-98, 128)	83 (-91, 4027)	-83 (-99, 124)	84 (-99, 68)
Firth of Thames	62	12	37	2	4.2	-31 (-92, 452)	104 (-84, 1508)	130 (-70, 1415)	-48 (-93, 277)	-83 (-98, 41)
Ohiwa Harbour	26	5	22	3	4.9	-64 (-81, -32)	-16 (-63, 81)	28 (-38, 153)	-75 (-85, -57)	-75 (-85, -57)
Ahuriri Estuary	25	11	18	3	4.8	216 (-81, 3659)	197 (-45, 1920)	-68 (-91, 24)	-73 (-93, 5.2)	233 (-76, 2963)
Farewell Spit	28	14	105	23	29.2	-67 (-85, -26)	-51 (-77, 0.49)	-24 (-59, 40)	-64 (-84, -21)	-66 (-85, -25)
Golden Bay	26	5	15	2	4.2	174 (-84, 4558)	-56 (-93, 97)	-89 (-98, -41)	-89 (-98, -41)	153 (-84, 3783)
Lake Ellesmere	24	11	659	317.5	319	-2.6 (-52, 101)	-2 (-43, 72)	-1.1 (-25, 33)	-2.1 (-44, 76)	-2.6 (-51, 100)

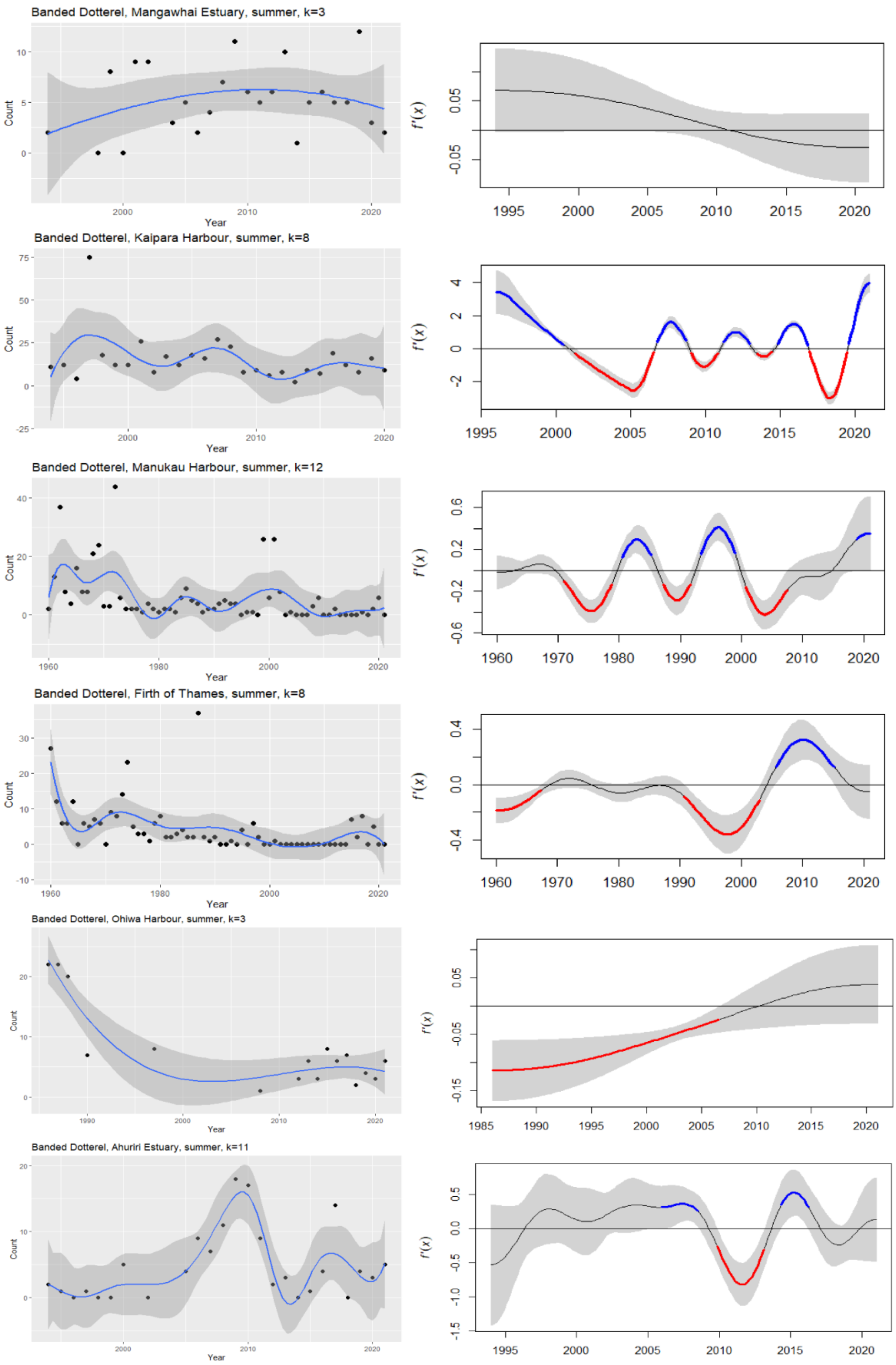

Figure 34. Site-specific trends of Banded Dotterels in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

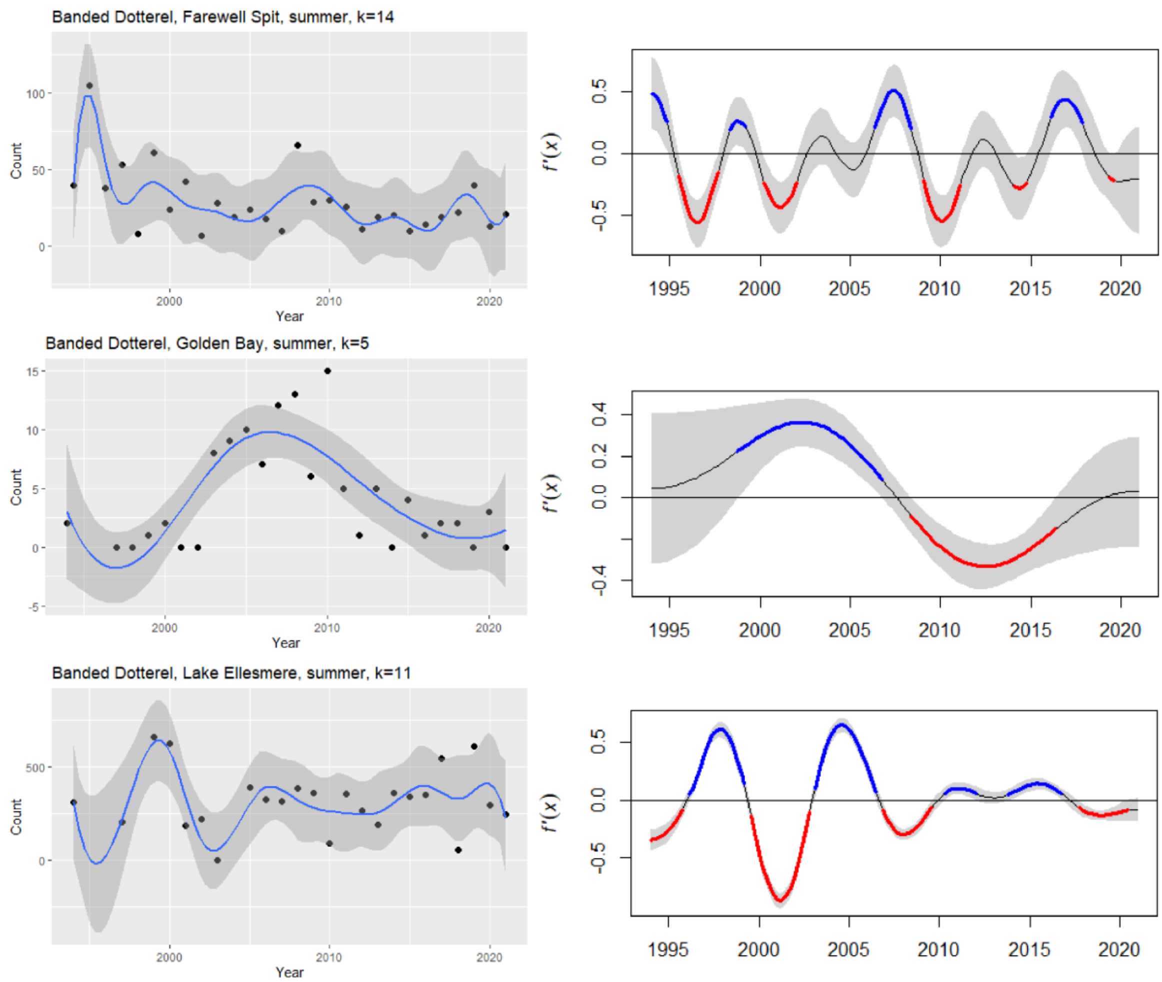

Figure 34 Continued

Table 29. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Banded Dotterels in summer. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Lat	0.34	Intercept	1261.14	1494.51	0.844	0.427
	$F_{1,7}$=0.5916, P=0.467, R^2=-0.0538		Latitude	-29.57	38.44	-0.769	0.467
2000s	Log(Area.ha)	0.94	Intercept	304.879	73.935	4.124	0.00444
	$F_{1,7}$=17.76, P=0.003967, R^2=0.6768		Log(Area. ha)	-38.982	9.251	-4.214	0.00397
2010s	Lat	0.34	Intercept	721.862	372.374	1.939	0.0937
	$F_{1,7}$=3.565, P=0.101, R^2=0.2428		Latitude	-18.085	9.579	-1.888	0.1010
MaxYr	MaxCount	0.49	Intercept	-66.59204	12.37084	-5.383	0.00103
	$F_{1,7}$=2.153, P=0.1857, R^2=0.126		MaxCount	0.07277	0.04959	1.467	0.18571

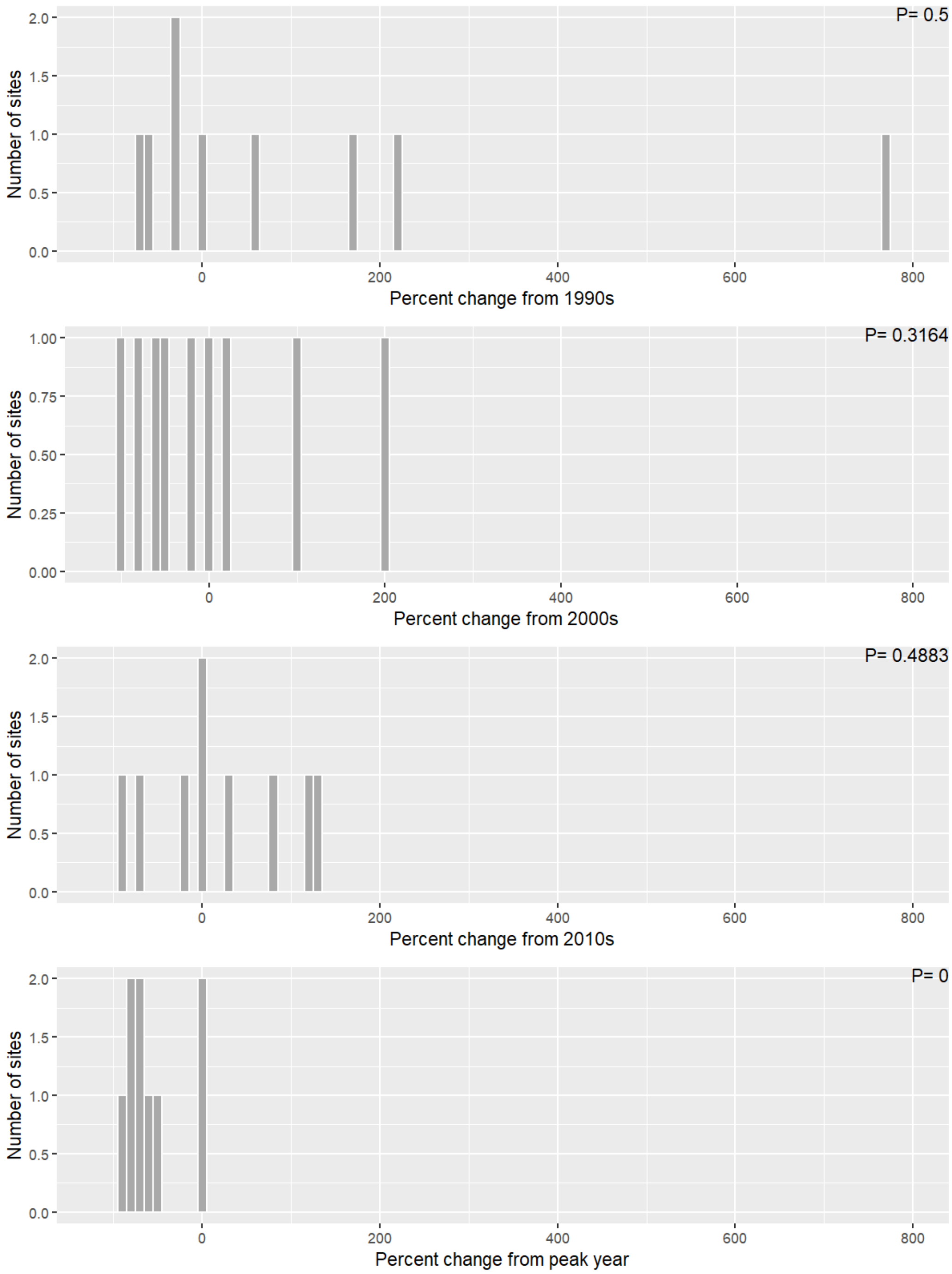

Figure 35. Individual site trends for Banded Dotterels in summer over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Wrybill

Winter

Modelled population trends for Wrybill in Winter varied from -98% to an increase of 406% depending on the site and time period (Table 30, Figure 36). The median trend was significantly lower than zero for all periods of the study, except the 1990s (median trends of -24% to -67%; significance values are given in Figure 37).

The large majority of the Wrybill population occurs in the Firth of Thames and Manukau Harbour. The Manukau Harbour population increased dramatically from 1000 birds to more than 3000 birds in five years from the late 1990s; this coincided with a decrease at the Firth of Thames from approximately 2800 birds to 1500 birds. This increased population in the Manukau Harbour has not lasted, with a decline through the 2000s and 2010s back to around 1000 birds. The Firth of Thames counts varied between c. 1500–3650 birds over the same period. Declines since 2000–2005 are also evident at Whangarei and Kaipara Harbours and the Manawatū Estuary.

Linear models suggest that population trends were positively affected by the size of the local population (Table 31). This result largely reflects the fact that two sites had very large counts and both had periods of increase during the surveys.

Table 30. Wrybill winter count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Whangarei Harbour	26	14	146	51.5	51.8	-98 (-100, -73)	-97 (-100, -61)	-96 (-100, -44)	-98 (-100, -68)	-98 (-100, -72)
Kaipara Harbour	27	15	308	161	164.2	-24 (-56, 42)	-42 (-64, -8.4)	-48 (-69, -19)	-51 (-71, -24)	-25 (-56, 37)
Waitemata Harbour	21	9	417	42	59.5	-78 (-96, 5.9)	-55 (-91, 111)	41 (-76, 621)	-67 (-93, 47)	-76 (-96, 11)
Manukau Harbour	61	15	4207	1330	1518.1	13 (-34, 89)	-42 (-64, -11)	-46 (-67, -11)	-48 (-67, -18)	-35 (-65, 11)
Firth of Thames	28	12	3650	2081	2115.9	-16 (-42, 17)	11 (-17, 52)	7.8 (-21, 55)	-5.2 (-22, 18)	-15 (-41, 17)
Tauranga Harbour	20	11	338	63.5	92	406 (-93, 60157)	-44 (-95, 558)	-90 (-99, -12)	-96 (-100, -66)	258 (-93, 25945)
Manawatū Estuary	58	15	71	23	26.5	-57 (-72, -29)	-71 (-82, -54)	-68 (-80, -49)	-71 (-82, -53)	-51 (-70, -8.1)

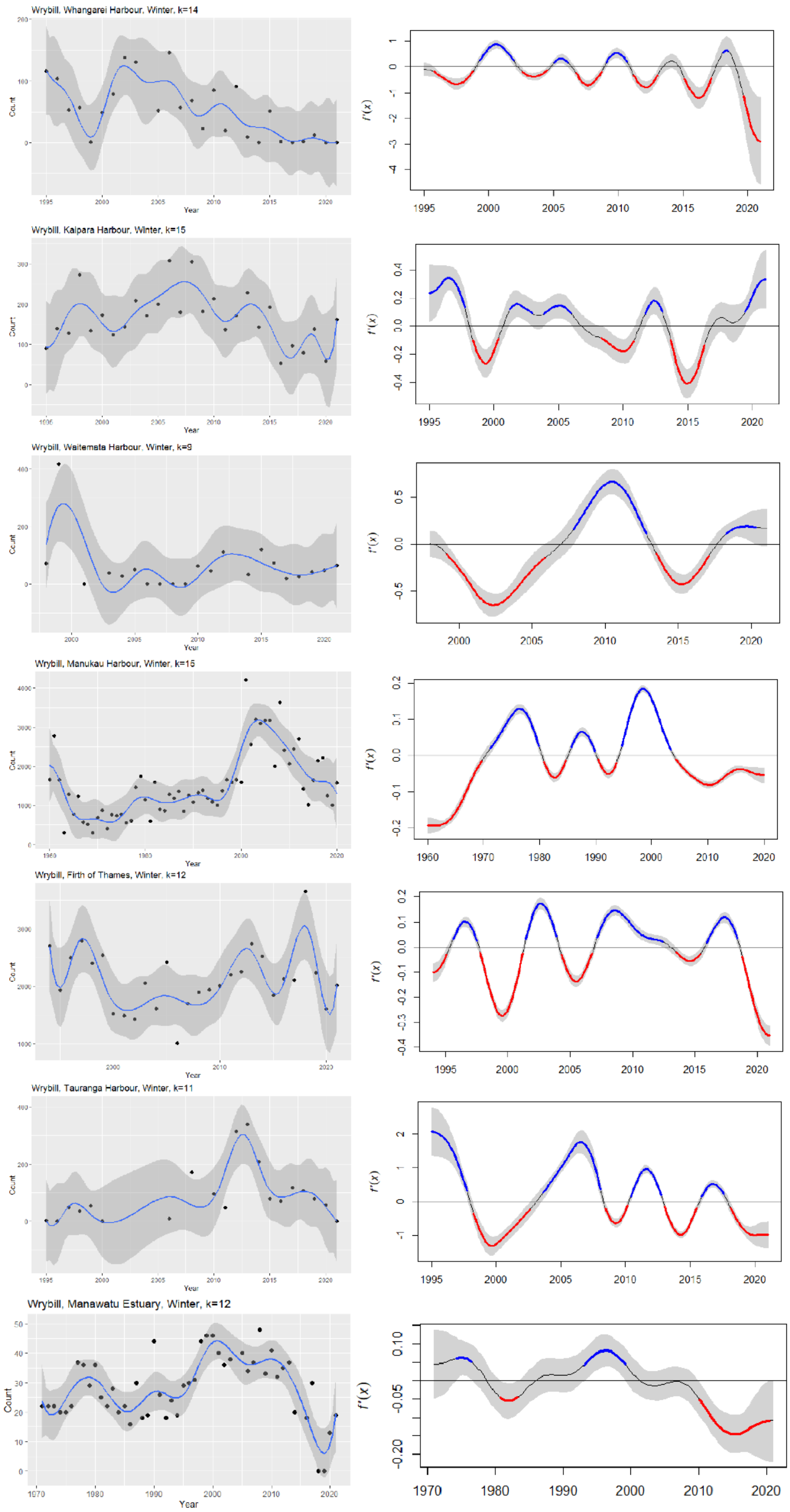

Figure 36. Site-specific trends of Wrybills in winter. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

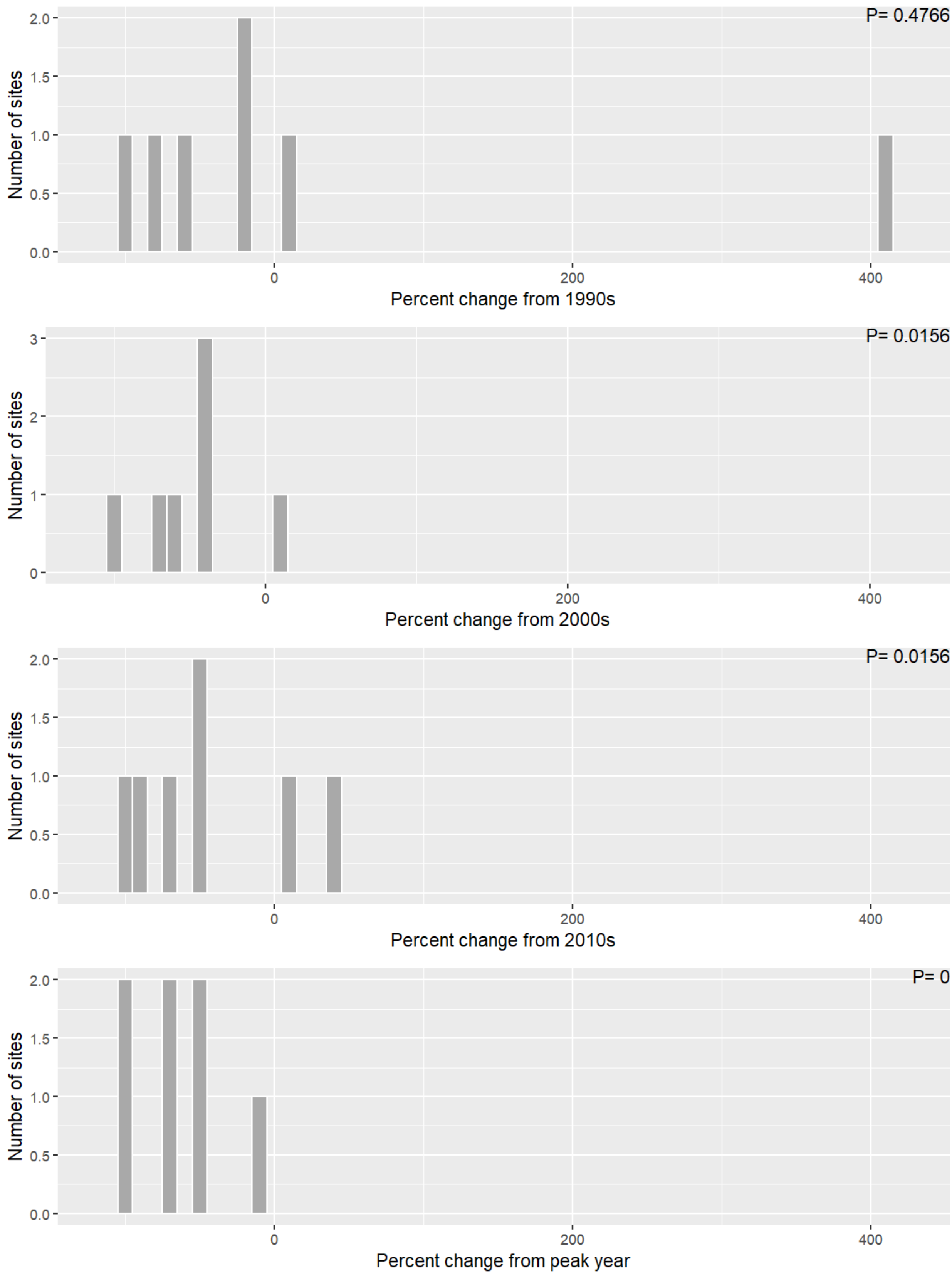

Figure 37. Individual site trends for Wrybills in winter over different analysis periods, with the significance value for the test of whether the median trend is different to zero shown in the right-hand top corner.

Table 31. Results of Linear Mixed Models testing individual site trends against maximum population size, area and latitude for Wrybills in winter. The model(s) with the best support (lowest AIC, or additional models within 2 AIC units of the best model) is shown. AICcWt shows the proportion of the total AIC variation that the specified model accounts for.

Time period	Model	AICcWt	Variable	Estimate	Std.err	t-value	P
1990s	Log(Area.ha)	0.47	Intercept	-250.20	293.02	-0.854	0.432
	$F_{1,5}$=0.9019, P=0.3859, R^2=-0.01662		Log(Area.ha)	33.19	34.95	0.950	0.386
2000s	MaxCount	0.75	Intercept	-64.400733	12.951705	-4.972	0.0042
	$F_{1,5}$=3.956, P=0.1034, R^2=0.3301		MaxCount	0.012158	0.006113	1.989	0.1034
2010s	MaxCount	0.42	Intercept	-54.664898	25.052729	-2.182	0.0809
	$F_{1,5}$=0.5998, P=0.4737, R^2=-0.07147		MaxCount	0.009157	0.011823	0.774	0.4737
MaxYr	MaxCount	0.88	Intercept	-79.167430	11.383653	-6.954	0.000945
	$F_{1,5}$=5.805, P=0.06091, R^2=0.4447		MaxCount	0.012944	0.005372	2.409	0.060914

Summer

Wrybill populations in Summer were modelled at only two sites, which varied from -36% to an increase of 60% depending on the site and time period (Table 32, Figure 38, Figure 39). Neither site holds many Wrybills at census time, as many young Wrybills migrate south for the breeding season (Dowding & Moore, 2006), but there is an indication that, at the Firth of Thames, over-summering numbers were higher in the 1960s and 1970s than in the 1980s and 1990s. Numbers have been relatively stable in the Manukau Harbour. Due to the small number of sites the linear mixed model analysis was not done.

Table 32. Wrybill summer count and trend summaries. Sites are ordered from north to south. The number of years of counts available is given, along with the number of splines (k value) in the Generalised Additive Mixed Model used to generate trend information. Maximum, median and mean counts in the time series are listed, followed by the trends (±95% confidence interval) to 2020/21 for periods from 1990, 2000, 2010, the year of the highest count, and the first year in the dataset.

Site	Years	k value	Max	Median	Mean	Trend 90s	Trend 00s	Trend 10s	Trend Max	Trend Yr1
Manukau Harbour	62	3	135	15.5	21.2	-36 (-72, 42)	-30 (-65, 42)	-19 (-50, 29)	-35 (-71, 43)	-15 (-68, 161)
Firth of Thames	62	14	312	63.5	87.7	50 (-33, 218)	60 (-28, 250)	21 (-40, 139)	-6 (-57, 87)	2.3 (-58, 151)

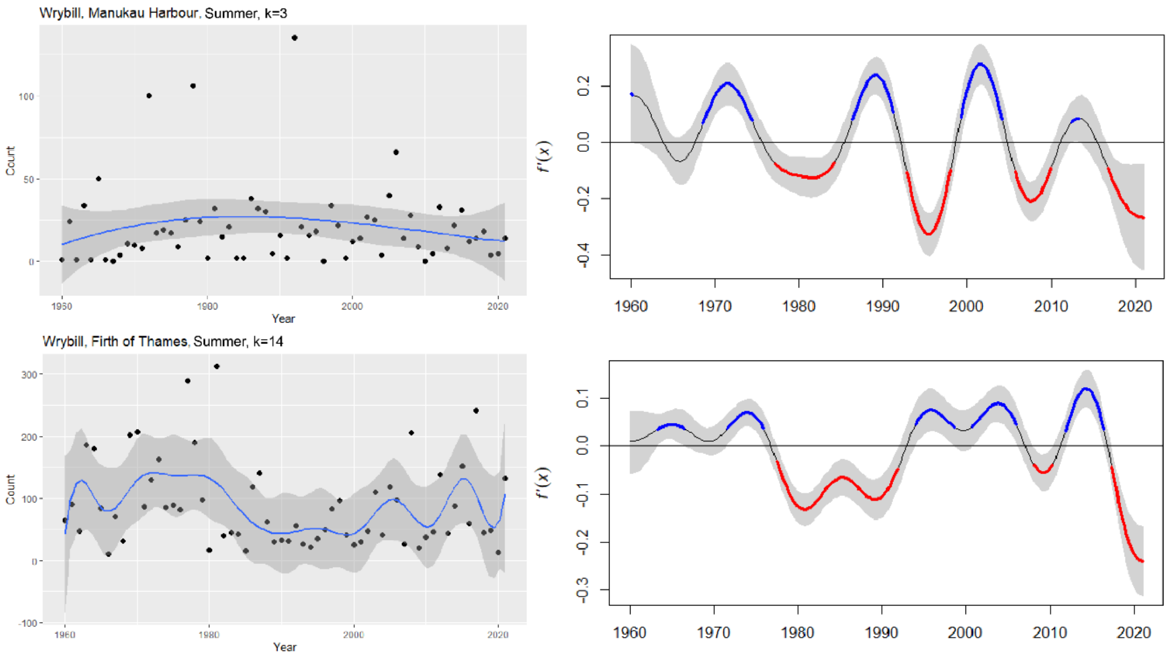

Figure 38. Site-specific trends of Wrybills in summer. Sites are ordered from north to south. Left-hand plots show the counts and modelled Generalised Additive Model trend with 95% confidence interval; titles give the number of splines (k) in the model. Right-hand plots show the relative trends, where red shows periods with a significant modelled decline and blue shows periods with a significant increase. Note that the x-axis scale varies between sites.

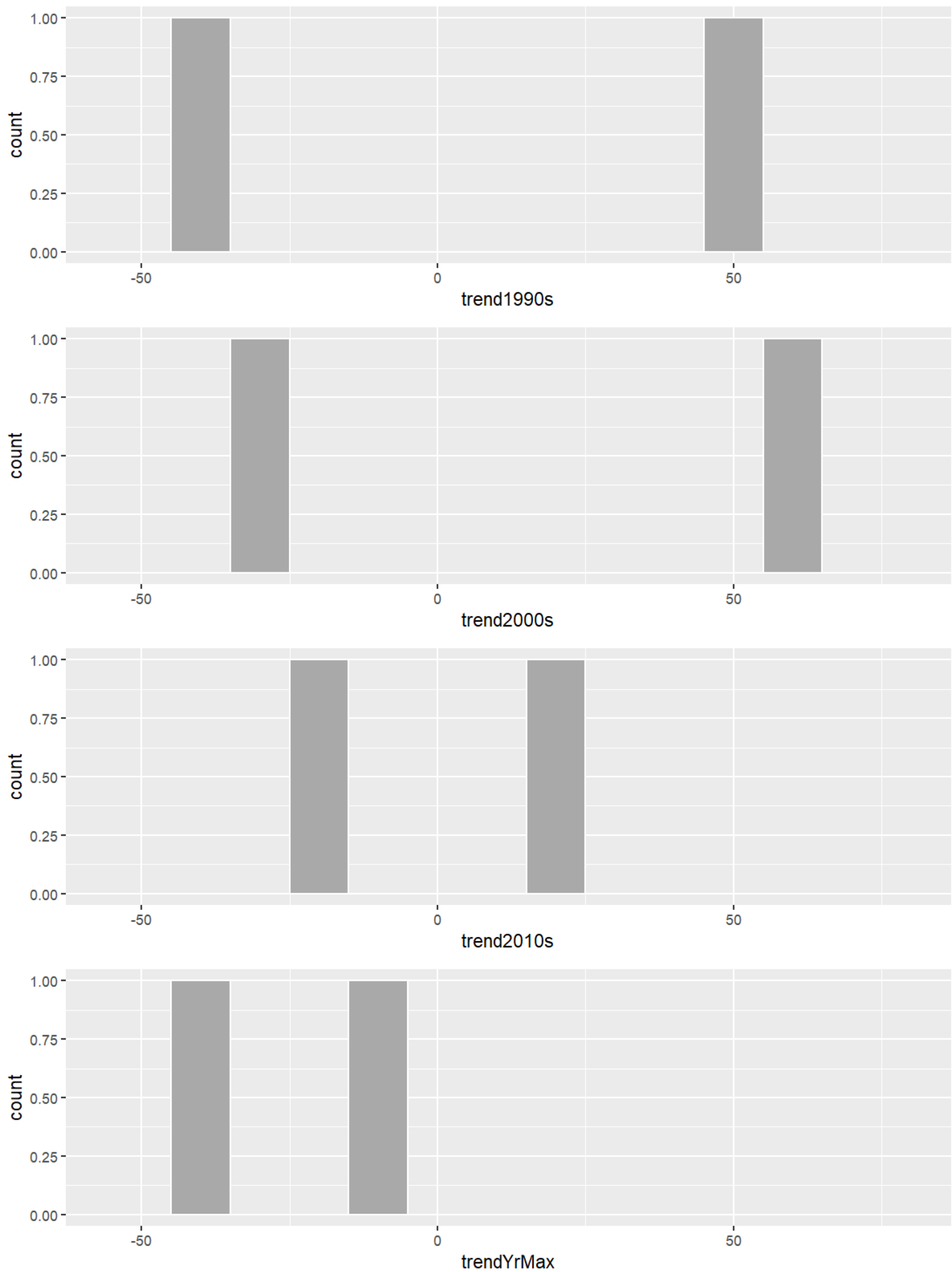

Figure 39. Individual site trends for Wrybills in summer over different analysis periods.

Chapter 4. Discussion:

This study confirms what many researchers were already aware of: most if not all shorebird species are declining across New Zealand, with practically all sites showing declines to some degree. The covariate tests indicate that some declines were seen to have been significantly affected by the latitude, area of the site and the population present at the site. This discussion looks to evaluate some of the main trends for each species and the possible reasons behind them. These population trends at the individual site level are driven by three main mechanisms—changes in survival, changes in recruitment and changes in site fidelity. Change in survival of adults is most obvious mechanism of a population decline at a site: if individuals do not survive then there will invariably be a decline. However, local population changes could also occur even without a decline in survival rate. This is where the other two mechanisms are at play. While recruitment is often used to represent individuals entering a breeding population, it can also represent the process of juvenile or immature birds settling at a site and therefore entering the local population. Declines in recruitment can lead to a declining population at a site despite a stable survival rate as with fewer new individuals entering the population, it will inevitably shrink. The last of the three mechanisms is site fidelity, which is how likely an individual is to return to the same site season after season. Adults of many species are highly site-faithful whereas juveniles and immatures may be far less site-faithful, and there may be differences between species in site fidelity of both adults and young birds. The local population will therefore reflect the balance of survival of all age-classes, recruitment of young birds, and site fidelity.

Arctic Migrants

Bar-tailed Godwits

Bar-tailed Godwits are the most prevalent of the arctic migrants found along the New Zealand shores, and therefore constituted the largest dataset in which to observe and analyse trends. Although most if not all summer sites are declining, there are several sites that display substantial sustained declines in population. Although having only a few census counts, Rangaunu Harbour in the Far North declined from early counts of 4000 birds to the latest count of 1400 birds, a 60% drop. Kaipara Harbour declined consistently from a peak of 21,823 birds in 2001 to 10,635 birds in 2020, a decline of

51%. Little Waihi Estuary had a decline of 73% from 1350 birds in 1998 to 363 birds in 2021, with the majority of the decline only occurring recently (61% in the 3 years since 2019). Ohiwa Harbour has been in near-constant decline since 1998, from 5000 birds to 1640 individuals in 2021, a decline of 67%. Kawhia Harbour declined constantly from 4353 birds in 2005 to the most recent count of 1650 birds in 2020, a decline of 62% over 15 years. Lastly, Manawatū Estuary has been in a constant decline since 1997 when it peaked at 570 individuals, to the most recent count of 154 birds in 2021, a decline of 73%. Clearly, godwit populations have declined widely around New Zealand.

The covariate trend analysis discovered a significant effect in the summer trends of latitude, in which sites further south have had more positive trends over time than have the northern sites. One possible explanation could be that differences in migration timing affect the fuelling conditions for southern and northern birds when on migration. Recent work monitoring migration timing found that Bar-tailed Godwits from southern New Zealand migrate northward to Asia on average 9.4–11 days earlier than godwits from northern New Zealand (Battley et al., 2020). Losses and deterioration of intertidal habitat around the Yellow Sea (Choi et al., 2015; Melville et al., 2016; Zhang et al., 2018) mean that there may be penalties for birds that arrive late (Peng et al., 2023): early-arriving individuals may have longer potential staging periods and face less food competition that birds that arrive later. In the New Zealand context, early migrating southern birds may have an advantage over later migrating northern birds. Increasing challenges in Asia are supported by data indicating that godwits from the Manawatū Estuary have started migrating to staging sites in Asia earlier in recent years, with advances in departure date of 6 days documented from 2008–2020 (Conklin et al., 2021). Despite this advancement in departure from New Zealand, birds did not reach Alaska earlier, suggesting that the shift in timing could be to compensate for deteriorating fuelling conditions in Asia. The continued decline in habitat quality at staging sites in the Yellow Sea area would put increased strain on late-migrating individuals in particular, by increasing competition for food resources as the season progresses and lead to reduced fuelling rates (Newton, 2006).

The covariate trend analysis for winter trends also suggested that latitude had an effect on the observed trends, though less so than the summer trends, with higher trends as latitude increased. This is likely driven by proportionately large population increases at far south sites in the most recent counts and the likes of Farewell Spit increasing from 1304 to 2626 birds in the span of four years. The overwintering component of the godwit population consists almost exclusively of young subadult birds, comprising juveniles (first-years) and immatures that could first migrate aged 2–4 years old (Battley et al., 2020). Individual marking work (Battley et al., 2020) and satellite tracking (J.R. Conklin and B. Kempenaers, unpubl. data) show that juveniles and immatures move extensively around New Zealand. It is possible that at some times, such as during years of high reproductive success when numbers of juveniles reaching New Zealand are high, large numbers of juveniles settle at the southern extent of mainland New Zealand. Due to their longevity and late breeding age, some individuals not breeding until their fifth season, the population would be expected to change slowly.

Insight into local site quality could come from comparing adult godwits, which are known to be very site-faithful between years (Battley et al., 2011; Conklin et al., 2021), and overwintering immatures that are known to be highly mobile. Adult numbers may be a poor indicator of site quality if they return to a site regardless of its current condition, whereas juveniles and immatures may sample one site yet settle at another, higher quality, site. High return rates of adults could combine with low recruitment to result in an overall population decline. The Manawatū Estuary may be an example of this. The population has declined continuously for almost 30 years, despite recent studies confirming high return rates of adults (Conklin et al., 2021), yet numbers of overwintering birds were too low to even enter the analyses.

The widespread declines in godwit counts within New Zealand implies that the population changes observed may be largely due to the global declines in population rather than being a New Zealand-centric phenomenon, as similar declines have been witnessed throughout sites in southeastern Australia as well ranging from reported annual declines of 3.22% from 1973–2014 (Clemens et al., 2016) when assessing trends on a continental scale. This magnitude of decline is corroborated by more site-specific analysis such as an annual decline of 2.4% for sites across Tasmania from 1974–2011 (Cooper et al., 2012) and annual declines of 5.86% from 1991–2009 for Western Point a site in South-eastern Australia (Hansen et al., 2015) as well as declines in population across other sites in South-eastern Australia (Schuckard et al., 2020). Periods of declining adult survival have been reported such as a drop from 0.91 to 0.84 from 2011–2012 (Conklin et al., 2016), which coincides with the initial period of a decline that was observed nearly nationwide in New Zealand. This decline in adult survival and hence decline in population is likely linked to the reclamation and deterioration of stopover habitats across the Yellow Sea area, such as Bohai Bay (Ma et al., 2014; Yang et al., 2011), Yangtze Estuary (Chen et al., 2016) in China and Saemangeum and Geum Estuaries (Lee et al., 2018; Moores et al., 2016) in South Korea among other sites (Studds et al., 2017). However, there are still some worrying occurrences in New Zealand, such as the disproportionately large decline at the Manawatū Estuary that presumably reflects local conditions as much as national trends. There were unprecedented high numbers of juvenile godwits in New Zealand in 2019 and more recently in 2022 and 2023 (P.F. Battley, pers. comm.). Such periods of high productivity should boost the national population for the next few years.

Red Knot

Most sites had declining summer counts, with only two sites showing an increase (Mangawhai Estuary in the 1990s and Rangaunu Harbour in the 2010s), and in both instances it was an increase from an initial low count. The largest population strongholds appear to have been stable in recent years, with Farewell Spit maintaining a population of roughly 7500 birds since the late 1990s and the Manukau Harbour maintaining roughly 10,000 birds since 2000, though it had virtually halved in number from the 1980s and 1990s.

The analysed trends did not show a significant effect of any of the covariates tested and so there were no systematic differences occurring between sites. With the observed trends being near-nationwide it is again likely that these trends result predominantly from factors occurring outside New Zealand. Red Knots have a heavy reliance on the stopover sites within the Yellow Sea area that have seen continued degradation across this time period (Chen et al., 2016; Lok et al., 2019; Rogers et al., 2010; Yang et al., 2011). The results display significant sustained declines across multiple sites during the 1990–2000 time period with Kaipara Harbour and Manukau Harbour being the worst affected, with declines of 5,222 and 10,112 birds respectively. Such declines have also been witnessed in South-East Australia, a noted decline occurring since 1992 of 5.5% which followed a period of significant increase in the area, and as such it is likely that the contributing factors to this decline are occurring on migration (Hansen et al., 2015).

Of the four sites analysed in winter, the three with smaller populations (Kaipara Harbour, Firth of Thames and Farewell Spit) displayed relatively stable populations with expected year-to-year variance. The largest population site, Manukau Harbour, increased since counts began around 1960 until the early 1990s, when the population of roughly 5000 birds at the site started declining over the ensuing decade to the now apparently stable population of approximately 1250 birds. Red Knots are known to be highly mobile between sites and have been observed to move between several sites during the same season, with banded birds being noted to move from the Firth of Thames to the Auckland Harbours during the winter period (Battley et al., 2011). The counts from the Manukau Harbour from 1990–2010 suggest a decreased number of juveniles and immatures, which does coincide with the witnessed decrease in the summer counts during the same time period.

Despite being the second-most numerous arctic migrant present in New Zealand and being highly mobile between sites, Red Knots are oddly absent from the Bay of Plenty and Waikato estuarine areas and harbours as well as the east coast of the South Island despite being found further south than that. It is not clear as to why these habitats are avoided as Red Knots primarily eat small bivalves that are present at most sites.

Studies in Australia have shown a decline in Red Knot populations as well, with a noted and distinct effect of latitude with the declines being worse in Southern Australia than in Northern Australia (Clemens et al., 2016). Trends at migration stopover sites have been also negative, not just at the particularly well-documented Bohai Bay in China (Lok et al., 2019; Rogers et al., 2010; Yang et al., 2011) but also, in South Korea (Moores et al., 2016). Though populations have been stable in Japan for many years, declines in Red Knot populations are now being reported there, though not as severe as elsewhere in the EAAF (Amano et al., 2010). As Red Knots traverse the Yellow Sea area twice per year (on both northward and southward migrations) instead of once a year like the aforementioned Bar-tailed Godwits (only on northward migration), we would expect impacts from issues in the Yellow Sea area stopover sites to be more severe in knots than in godwits.

Ruddy Turnstone

Trends of Ruddy Turnstone counts were mixed, and there was no systematic increase or decline across the surveyed sites in recent years. The two observed winter sites have shown an increase in population while most of the summer sites displayed a relatively stable population. This apparent stability may indicate a hopeful future for the species in New Zealand, following what appears to have been a poor situation since the early 1990s, when the long-term sites of Manukau Harbour and Firth of Thames experienced their most significant declines of 85.6% from 1992 to 2000, and 77.5% from 1991 to 1995, respectively.

While there were no significant correlates of trends across sites, individual trends indicate that local site conditions are playing a role in the population changes. The neighbouring sites of the Firth of Thames and Manukau Harbour have shown opposing trends, with the Firth of Thames declining to very low numbers while the Manukau Harbour population remained stable since 2000. Awarua Bay declined to a low stable population since 2005 while the neighbouring Invercargill had an increasing population in the same period. Despite experiencing a decline in the summer season, winter counts at Awarua Bay have increased over the past decade.

The declines through this time period are in alignment with declines witnessed in Australia with Ruddy Turnstone numbers in Tasmania being in decline since 1974 (Cooper et al., 2012). The overall population of Ruddy Turnstones across New Zealand has likely declined, with historic counts estimating the nationwide population at approximately 5000 birds in the 1980s (Sagar et al., 1999) while the nationwide population has been more recently estimated at 1654 birds (Riegen & Sagar, 2020).

Pacific Golden Plover

The Pacific Golden Plover has been found to be in a national decline, with declines at the Firth of Thames from a maximum count of 246 to the most recent count of 20, a decline of 91.8%. This scale of decline is witnessed across New Zealand, with the covariate analysis identifying the importance of area of the habitat to the species, with a positive relationship between trend and area in three of the analysed time periods. This means that although the population is declining nationally, the decline is lessened at the larger sites across New Zealand. This may represent a buffer effect with the populations retracting to these larger sites such as Kaipara Harbour, Farewell Spit and Manukau Harbour. These sites have managed to maintain relatively stable populations while others have had continued drastic declines, such as at the Manawatū Estuary that has been in constant decline since the 1980s from a maximum of 46 birds to the most recent survey of three birds, a decline of 93.4%. Unless things change, the species will eventually be lost at the Manawatū Estuary, and the same fate is likely at other monitored sites including Houhora Harbour, Little Waihi estuary, Ohiwa Harbour and Lake Wairarapa.

Older counts from 1983–1994 have the total Pacific Golden Plovers across the country estimated at 649 birds (Sagar et al., 1999) which has declined to the total number of birds present in this analysed dataset in the most recent year to 53 birds, a nationwide decline of 91.8% across four decades. Studies conducted in Australia are also showing declines in this timeframe, though not as drastic as the declines being observed in New Zealand, with sites from both Western Australia and New South Wales showing declines ranging from 1% to 63.8% depending on the site (Hansen et al., 2015; Rohweder, 2007). Migration data gathered by the Miranda Shorebird Centre has found that Pacific Golden Plovers departing from the Firth of Thames primarily do not migrate through the Yellow Sea Area but instead migrate through Guam and Japan to reach the breeding habitats of Alaska and as such we would expect their decline to be less than that of the other arctic migrants that utilise the Yellow Sea stopover areas if the declines were solely caused by the deterioration in the Yellow Sea area. However, the decline experienced by the Pacific Golden Plovers is of an equivalent or greater magnitude than the other arctic migrants. Local conditions may play a role in declining trends at some sites, such as the southern shores of the Firth of Thames where mangroves have encroached on saltmarsh areas formerly used by Pacific Golden Plovers. The size of the declines in New Zealand, and how widespread they are, nevertheless imply that the main causes of decline occur overseas. Stopover conditions could be declining at other sites away from the Yellow Sea, and as New Zealand is at the southern end of the range of the Pacific Golden Plover it is possible that the population as a whole is retracting away from New Zealand with even modest global population changes.

Red-necked Stint

Red-necked Stints have declined at most sites monitored and are now present in only low numbers at the northern sites. The largest numbers traditionally have been in southern sites (Lake Ellesmere and Awarua Bay) (Sagar et al., 1999), but even these have declined. At Lake Ellesmere, numbers declined by 82% over a decade from 2000 to 2010, yet the long-term decline is even larger than this. Older counts not included in this study recorded 220 birds in 1980 (O'Donnell, 1985), making it a decline of 90.9% in four decades. Concerningly, the rate of decline was evidently much higher from 2000–2010 (8.2% per annum) than across the whole period from 1980 onwards (2.6%). The winter counts at the Manukau Harbour suggest that the number of juvenile Red-necked Stints that reach New Zealand is likely variable and the two peaks present in the database are likely periodic increases arising from good breeding success. Both winter sites have their population peaks approximately mirrored in the following summer seasons at Manukau Harbour and Lake Ellesmere.

Red-necked Stints are one of the most numerous species of migratory shorebird in Australia (Clemens et al., 2016) and, indeed, in the East Asian-Australasian Flyway (Hansen et al., 2022), but analyses of counts in Australia indicate that the species is slowly declining (Clemens et al., 2016; Studds et al., 2017). Within Australia there have been geographical differences in trends, with populations declining faster in the southern half of country than in the north. New Zealand is at the extreme south-eastern edge of the range of Red-necked Stints, and the New Zealand population effectively consists of spillover from Australia (Mu et al., 2020). The analysis of Studds et al. (2017) suggested that the rate of decline in Australia increased between 2000 and 2005. In Japan, stint populations on migration declined faster in the decade up to 2008 than in the previous two decades (Amano et al., 2010). This is similar to the trends in the Manukau Harbour and at Lake Ellesmere, indicating a common cause of decline such as the coastal degradation of the stopover sites in the Yellow Sea area more so than that of internal factors within New Zealand. While this decline in the Yellow Sea area has undoubtedly affected Red-necked Stints negatively they have been found to be somewhat resistant to the degradation occurring throughout the area in comparison to other similar species such as the Curlew Sandpiper, which declined so much that there were insufficient data for analysis in this study. The comparatively better population trends of stints than Curlew Sandpipers is thought to be due to their use of many different stops throughout the migration and Yellow Sea area rather than a limited number of key sites (Lisovski et al., 2021).

Sharp-tailed Sandpiper

The Sharp-tailed Sandpiper is a migratory shorebird for which New Zealand is at the edge of its known migratory range. The largest number recorded in national censuses in this dataset was 60 in 2005, with the highest individual count being 40 at the Firth of Thames in 1985. Considering that the earliest counts did not include all sites that now are in the modern counts, it is highly likely that the population has been in a nationwide decline since the mid-1980s, if not earlier. It is likely that a large portion of the decline in Sharp-tailed Sandpipers can be attributed to the continued deterioration of the stopover habitat in the Yellow Sea that the birds rely on for their migration rather than internal factors within New Zealand as the decline has been constant in pace and experienced in sites nationwide. The Sharp-tailed Sandpiper had the third-highest rate of decline in Clemens et al.'s (2016) analysis, with decreases in the south of Australia but increasing or stable populations in the north. It would seem that the species is either retracting towards the north of Australia or is shifting northwards. Either way, these findings fit with the observed declines in New Zealand.

Internal Migrants

South Island Pied Oystercatcher

The South Island Pied Oystercatcher expanded its breeding range in the 20th Century away from the Canterbury braided riverbeds, and it now breeds throughout the South Island and in small numbers on riverbeds of the lower North Island. This expansion came after hunting was made illegal in 1940 (Baker, 1973), and involved shifts into agricultural habitat. There was a well-documented and massive increase in numbers around the Auckland region (Veitch & Habraken, 1999) with the estimated national population going from 10,000 birds in the 1940s to 49,000 birds in the 1970s (Baker,

1973) to 112,000 birds in the early 1990s (Sagar et al., 1999). The population is thought to have declined since then, with only 79,000 birds estimated in 2019 (Riegen & Sagar, 2020). Predominantly southern breeding birds may have been adversely affected by agricultural changes with a significant proportion of sheep farms that were used as breeding grounds having switched to dairy farms, which the species has struggled to utilize as breeding habitat (Riegen & Sagar, 2020). The witnessed declines throughout the winter sites are likely linked to poor juvenile recruitment to the sites.

There have been widespread increases across sites in the summer season since 2010 with sites like Lake Ellesmere increasing from its 2010 population of 3 birds to the population peak of 98 birds in 2020, an increase of 3260%, with the data indicating there may be a cyclical effect as similar increases though of a smaller magnitude have been identified at roughly 10-year intervals. The linear models of the covariate analysis found both latitude and area had significant positive effects on the population.

In the Winter season, however, the covariate trend analysis found that both latitude and area had significant negative effects on the observed trends. These likely arise from northern sites having more modest decreases than southern sites like Farewell Spit, Avon-Heathcote Estuary and Otago Harbour (which had impactful declines of 58.8%, 78.8% and 63.2% respectively), or increasing in some periods (Waikato and Bay of Plenty sites).

Baker (1973) suggested that South Island Pied Oystercatchers would eventually become density-dependent if they continued to increase in number. It is possible that the decrease in the total population, which is likely related to changes on the breeding grounds, has relaxed any density-dependent impacts at the large non-breeding sites in northern New Zealand. This would allow birds to preferentially winter at these northern sites, where the warmer environments would reduce energetic costs.

The adult South Island Pied Oystercatchers have been found to have a high site-fidelity for their wintering habitats with records of them returning year after year (Sagar & Geddes, 1999). It has been noted that over time migration between wintering and breeding sites has become earlier, while the timing of counts occurring has not. This means that the counted numbers of South Island Pied Oystercatchers may have been underestimated and that the indicated declines in the trends may be less severe than the data indicates.

Pied Stilt

Trends of Pied Stilts were variable over time and between sites, and there were no common patterns or relationships with site variables in either winter or summer counts. Numbers of stilts at sites in summer are likely to represent mainly local breeders, whereas counts in winter potentially include migrants from inland and from further south in New Zealand. Most of the winter sites with smaller populations have experienced a slight growth in population since the dataset record started, but some of the largest populations have experienced major declines. The Firth of Thames, for instance, has been previously identified as the most important non-breeding site in New Zealand for Pied Stilts (Dowding & Moore, 2006) but from its peak of 6049 birds in 1996 it declined by 54% over the next three years. The population stabilized over the next decade before undergoing another decline from 5111 birds to 1990 birds over the span of four years from 2010 to 2014, a decline of 61%.

In comparison most of the summer site observations display a decline in recent years with Whangarei Harbour, Kaipara Harbour, Manukau Harbour and the Firth of Thames displaying significant declines in the north while Lake Ellesmere has displayed a significant decline in the south. As the summer populations are thought to consist of breeders and early flockers it is likely that although the trends are not constant across the sites in terms of timing or magnitude, these trends showing significant population change are indicative of changes in local conditions. The covariate analysis did not find any significant associations with the area, population size or latitude, but other local factors are likely important in sustaining populations. Pied Stilts often breed in flooded or wet paddocks away from coasts (Dowding & Moore, 2006), and changes in agriculture (active draining of farmland) and climate (rainfall patterns and drought) make it likely that much of the variation in stilt counts reflects changes in breeding ground conditions.

Banded Dotterel

The analysis of the trends for Banded Dotterels found no covariate played a significant role in the trend of the population for the winter season. The summer season is not well surveyed as most individuals breed inland away from the coast. However, those that do breed on the coasts face an increased risk from human disturbance and predators.

Banded Dotterels can be difficult to survey accurately, as they often utilise fields away from the shore, and there is large uncertainty about the true population of Banded Dotterels. A substantial proportion of the population migrates to Australia for the winter, with most of these migrants being southern inland-breeding birds as opposed to coastal-breeding birds (Pierce, 1999). One estimate was that 74% of the inland-breeding population winters outside New Zealand (Pierce, 1999), and it is possible that there is a significant population trend that is being obscured by the lack of available data.

Studies of banded birds indicate that birds counted at northern sites are likely to be coastal-breeding dotterels whereas those found at southern sites may be inland-breeding birds (Pierce, 1999). This is concerning as Lake Ellesmere, which has historically been the largest stronghold of the southern sites, has been in a constant decline since its peak in 1995 of 2168 birds to the most recent count of 574 birds, a decline of 73.5%. This provides a worrying insight into the state of the inland breeding populations on the braided rivers of the South Island that are thought to constitute the majority of birds at Lake Ellesmere. Historic counts from the Firth of Thames not included in this dataset indicate that a large decline in Banded Dotterel has likely already occurred. Counts in the 1950 winter season recorded 2000 birds (Fleming & Stidolph, 1951) while the most recent winter season count in 2021 totalled only 189 birds. Even at the start of this dataset in 1994 the population had already been reduced to 160 birds being counted, a loss of 92% across the 44 years.

The decline at the Firth of Thames is likely an effect of the habitat loss occurring through continued mangrove expansion (Swales et al., 2007). This has completely covered the southern coastline of the Firth which is where historically the largest counts have occurred. This has both removed habitat from the upper tidal flats and saltmarsh boundary, presumably limited foraging opportunities as well as pushing birds into closer proximity to the mammalian predators (feral cats, rodents and mustelids) that have been damaging bird populations since being introduced to New Zealand (Dowding & Murphy, 2001).

The dotterel population appears to be experiencing a slight resurgence, with recent population increases at both Firth of Thames and Manukau Harbour during the winter season. This is good news as both the Firth of Thames and Manukau Harbour had witnessed an extended period with no counts witnessing them, implying the species may have been lost from those sites. This evidence indicates that the sites are likely only utilised as a wintering ground in recent years.

Wrybill

Wrybills are the most restricted species in terms of sites used, with the bulk of the population being at just two sites. This means there is limited scope to detect effects of site factors, and the one that was found (a weak effect of maximum count on trends) is somewhat uninformative. The most important finding is that the apparent shift of much of the population from the Firth of Thames to the Manukau Harbour has not been permanent.

With Wrybills being heavily restricted in terms of breeding habitat, they are more adversely affected by extreme events such as flooding of the braided rivers that they breed upon. Studies have found a strong negative correlation between the peak instantaneous discharge of the Rakaia River, one of the braided rivers Wrybill are known to breed along, and the summer counts of (young) Wrybill at northern harbours the following season (Hughey, 1985). This implies that the Wrybill numbers observed at northern harbours are heavily influenced by extreme weather events occurring on the breeding grounds. This is concerning, as the number of extreme weather events is increasing alongside climate change (Stott, 2016). In seasons where birds are dislodged from the breeding habitats, the importance of neighbouring sites such as Lake Ellesmere that do not usually see much use during the breeding season rises, as they are used for refuge (Crossland & Crutchley, 2020). If sites used for refuge are not of a sufficient quality it may have compounding effects as the aftereffects of a flood leave the breeding habitats along the braided rivers scarce on food (Crossland & Crutchley, 2020).

As the Wrybill breeds exclusively in the South Island of New Zealand in difficult-to-survey places, population monitoring can only be done on the nonbreeding grounds. The summer counts at northern sites should consist entirely of non-breeding juveniles and provide a clear indicator of the previous breeding season's productivity, though there are records of one-year old birds that migrate to the breeding grounds with the adults (Dowding & Moore, 2006). There are significant declines during the winter season at both Whangarei and Kaipara Harbours since 2005 with declines of 100% and 71% from 2005–2020 respectively.

Manukau Harbour experienced a population increase at the start of the 2000s which is believed to be in part due to the closure of the old oxidation pools used by the Mangere Wastewater treatment plant. These ponds were progressively returned to tidal flats between 1998 and 2003 (Papps & Priestley, 2003), providing a rich coastal habitat with silty mud, the preferred soft-sediment foraging habitat for Wrybills (Withington, 2015). Wrybills are known to move between the sites (Dowding & Moore, 2006) and it appears that birds quickly responded to the habitat changes in the Manukau Harbour. The fact that the population seems to have returned to the Firth of Thames implies that these benefits may have diminished over time.

At the other end of the population scale, the Manawatu Estuary more than halved since 2000. In absolute numbers this decline is fairly trivial (~20 birds) and is likely to be indicative of a more systematic local change, such as a change in the macrobenthos present at the site.

Analysis Overview

A key aim of this thesis was to test for evidence of a buffer effect operating in shorebird populations in New Zealand, in which populations might 'retreat' into higher quality sites, with lower-quality sites having greater rates of decline. Without direct measures of species-specific site quality, maximum population size, intertidal area and latitude were included as proxies for potential quality. Significant associations of these tests across species are summarised in Table 33. Across all the multiple tests of one or two seasons for eight shorebird species, maximum population size was only significantly associated with a population trend once (for Pacific Golden Plover), and marginally so twice. Latitude was more strongly associated with trends in godwits (better trends in the south) and oystercatchers (better trends in the north). Site area had three significant associations (positively for Pacific Golden Plovers and South Island Pied Oystercatchers in summer, but negatively for oystercatchers in winter). Overall, there is little evidence of a buffer effect operating, at least with the inputs in this study. For northern hemisphere breeders it is likely that the majority of the population drivers are occurring overseas. It is possible that the steep declines in less common Arctic migrants are an indication of a buffer effect occurring at large scales, with New Zealand as a whole being less favoured habitat. The one case where a species may be retreating to better sites in New Zealand is the Pacific Golden Plover. By looking at the summary of site trends shown in Table 34 it can be clearly seen that arctic migrant species have fared worse over the analysed time periods than that of internal migrants with the comparison of significantly negative time periods being 14 for arctic migrants and 8 for internal migrants. Table 34 also shows that in all but two of the combinations there is a significant decline from the peak count and the two that aren't declining are near significant, indicating that all species are actively declining in the present day.

The New Zealand species showed little to no indication of a buffer effect occurring. The trends of the internal migrants likely reflect variations arising from local conditions such as changes in breeding conditions or local habitat quality. There appears to be a geographic difference in the rate of population change in the South Island Pied Oystercatcher population.

*Table 33. Summary of Covariate analysis tests carried out, with significant effects being denoted by * and near-significant effects being denoted by (*). A dash indicates that no covariate was significant, the number of asterisks shows how many of the tests were significant.*

Species	Season	MaxCount	Latitude	Area
Bar-tailed Godwit	Summer	-	***(*)	-
	Winter	-	(**)	-
Red Knot	Summer	-	-	-
Ruddy Turnstone	Summer	-	-	-
Pacific Golden Plover	Summer	*(*)	-	*
South Island Pied Oystercatcher	Winter	(*)	*(*)	*
	Summer	-	*(**)	**
Pied Stilt	Winter	-	-	-

	Summer	-	-	-
Banded Dotterel	Winter	-	-	-
	Summer	-	-	*
Wrybill	Winter	(*)	-	-

Table 34. Summary of tests of whether median counts were significantly different to zero for the analysed species/seasons combinations. Ns means not significantly different to zero, (-ve) means nearly significantly different to zero in a negative direction, and -ve means significantly more negative than zero

Group	Species	Season	1990s	2000s	2010s	PeakYr
Arctic migrants	Bar-tailed Godwit	Summer	-ve	-ve	(-ve)	-ve
		Winter	ns	ns	ns	(-ve)
	Red Knot	Summer	ns	-ve	-ve	-ve
		Winter	-ve	ns	ns	-ve
	Ruddy Turnstone	Summer	ns	ns	ns	-ve
	Pacific Golden Plover	Summer	(-ve)	(-ve)	(-ve)	-ve
	Red-necked Stint	Summer	ns	-ve	-ve	-ve
	Sharp-tailed Sandpiper	Summer	ns	ns	ns	-ve
New Zealand migrants	South Island Pied Oystercatcher	Winter	ns	ns	ns	-ve
		Summer	ns	ns	ns	-ve
	Pied Stilt	Winter	ns	ns	ns	-ve
		Summer	ns	ns	ns	-ve
	Banded Dotterel	Winter	ns	ns	ns	(-ve)
		Summer	ns	ns	ns	-ve
	Wrybill	Winter	ns	-ve	-ve	-ve

Conclusion:

In conclusion the study has shown that populations of migratory shorebirds have been in decline across nearly all of New Zealand's sites. Though some of the larger strongholds are declining slower than most of the other sites (Firth of Thames, Manukau Harbour) they are still at numbers that are reduced by up to 95% in some places from what they were. This is likely to have arisen from multiple contributing factors elsewhere on the flyway as the covariate analysis displayed that the assessed covariates had little effect on the observed trends as most covariate/species/site combinations were non-significant. The internal migrant species had more positive results with both Pied Stilts and South Island Pied Oystercatcher having several sites where they are displaying a trend of increasing population.

This study provides an exploratory insight into the population of shorebirds within New Zealand and provides evidence for the need for continued monitoring efforts throughout New Zealand. Given national and global environmental changes further research into the specifics of trends observed here is warranted to understand the cause behind the witnessed declines at the local level.

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