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Changes in the bird community of Auckland Domain's urban forest between 1987 and 2020

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Abstract: The Auckland Domain is the city's oldest park and contains over 70 ha of contiguous, mature urban forest. Five-minute bird counts were made across one year within the domain forest in 2019 and 2020 and compared with counts conducted in 1987 and 1988, using the same methods and at the same survey sites, to investigate changes in the structure of the urban bird community. The abundance and species richness of native and introduced birds increased between the count years and there was structural change within the community driven by increases in the abundance of forest-adapted endemic species, tūī *Prosthemadera novaeseelandiae*, grey warbler *Gerygone igata*, and kererū *Hemiphaga novaeseelandiae*, and declines in generalist native species, silvereye *Zosterops lateralis* and fantail *Rhipidura fuliginosa*. Tūī showed the most profound increase in abundance between count years, reflecting regional conservation management of mainland and island forest habitats that benefit this highly mobile species. Increased abundance of eastern rosella *Platycercus eximius* and common myna *Acridotheres tristis* also altered community structure between count years, indicative of ongoing colonisation by these exotic species in the Auckland region since their introduction to the North Island. The fact that both these species compete with native taxa for nest cavities within forests is of concern. Our results reinforce the need to manage and protect maturing urban forests to enhance native bird populations. Such actions will also support the recovery of native bird populations at a landscape scale.

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Key words: Five-minute bird count, species abundance and diversity, pest control, forest succession

INTRODUCTION

New Zealand's native forest bird populations have been significantly impacted by human settlement through loss of habitat, increased competition with introduced birds, and the catastrophic impact of introduced mammalian predators and herbivores (Krull *et al.* 2015). These impacts have seen the extinction of some species and the retreat of others to remote forest habitats in protected mainland and offshore islands (Diamond 1984; Tennyson & Martinson 2007), though a suite of more adaptable species have maintained populations in human modified forested landscapes (Miskelly 2018; Fitzgerald *et al.* 2019). One of the more challenging and interesting of these habitats are highly modified urban 'forests' constituting either a matrix of backyard emergent vegetation or remnant isolated pockets of mixed native and exotic vegetation within the urban landscape. How forest bird communities respond to the increasing impacts of urbanisation is of interest in the field of urban ecology (Galbraith *et al.* 2015).

Auckland, with over 1.6 million people, is New Zealand's largest urban centre, and is experiencing rapid urban population growth. For example, between 1980s and today the city's central business district (CBD) increased from approximately 2000 residents to over 50,000 and housing intensification and relaxed tree protection laws have seen the removal of much urban forest habitat (Wyse *et al.* 2015). On the edge of Auckland's CBD, the Auckland Domain (275 ha) is the city's oldest park,

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Table 1. Species recorded during 5-minute point counts conducted within urban forest patches in 1987–1988 and repeated in 2019–2020, in Auckland Domain, Auckland, New Zealand. Species are listed in order of mean abundance across all counts (*N* = 379). An indication of which species have been included for each analysis is also given. Species shaded in grey showed significant differences in abundance between survey years in GLMM models.

	Species	Scientific name	Primary diet†	Occupancy (%)			Mean abundance			Overall richness measures	Community analysis	Species response GLMMs
				1987/88	2019/20	Overall	1987/88	2019/20	Overall			
1	Silvereye	*Zosterops lateralis*	I, F	91.80	71.74	82.06	3.19	2.364	2.789	X	X	X
2	Eurasian blackbird*	*Turdus merula*	G	78.46	88.04	83.11	1.559	2.179	1.86	X	X	X
3	Tūī	*Prosthemadera novaeseelandiae*	O	21.54	90.22	54.88	0.236	2.663	1.414	X	X	X
4	New Zealand fantail	*Rhipidura fuliginosa*	I, F, N	83.08	67.94	75.73	1.415	1.299	1.359	X	X	X
5	Grey warbler	*Gerygone igata*	N	43.59	64.13	53.56	0.518	1.038	0.77	X	X	X
6	Eastern rosella*	*Platycercus eximius*	I, F	1.03	54.89	27.18	0.01	1.109	0.544	X	X	X
7	Chaffinch*	*Fringilla coelebs*	G	32.82	33.70	33.25	0.497	0.511	0.504	X	X	X
8	Song thrush*	*Turdus philomelos*	I, F	22.05	23.91	22.96	0.282	0.326	0.303	X	X	X
9	House sparrow*	*Passer domesticus*	I	3.59	23.91	13.46	0.056	0.424	0.235	X	X	X
10	Goldfinch*	*Carduelis carduelis*	G, F, H	7.18	22.28	14.51	0.092	0.359	0.222	X	X	X
11	Greenfinch*	*Chloris chloris*	G, I, F	15.39	7.61	11.61	0.272	0.087	0.182	X	X	X
12	New Zealand kingfisher	*Todiramphus sancta*	I	16.41	12.50	14.51	0.185	0.174	0.179	X	X	X
13	Common myna*	*Acridotheres tristis*	I	2.05	13.59	7.65	0.021	0.217	0.116	X	X	X
14	Common starling*	*Sturnus vulgaris*	G	6.67	4.35	5.54	0.087	0.054	0.071	X	X	X
15	New Zealand pigeon	*Hemiphaga novaeseelandiae*	I, C	0.51	3.80	2.11	0.01	0.043	0.026	X	X	X
16	Rock pigeon*	*Columba livia*	C, O	1.03	2.17	1.58	0.01	0.027	0.018	X		
17	Welcome swallow	*Hirundo neoxena*	G	0.00	2.72	1.32	0	0.027	0.013	X		
18	Spotted dove*	*Streptopelia chinensis*	I, C	0.00	1.63	0.79	0	0.016	0.008	X		
19	Australian magpie*	*Gymnorhina tibicen*	I	0.00	1.09	0.53	0	0.011	0.005	X		
20	Shining cuckoo	*Chrysococcyx lucidus*	I, O	0.51	0.54	0.53	0.005	0.005	0.005	X		

* Introduced species. † Primary dietary component/s (derived from Heather & Robertson 1996): C = carnivore (vertebrate prey), F = frugivore, G = granivore, H = herbivore, I = insectivore (insect and invertebrate prey), M = molluscivore, N = nectarivore, O = omnivore (broad diet which may include invertebrates, lizards, chicks, eggs, carrion, fruit, seeds, refuse or waste). NB: main dietary component listed first where two or more components given.

set aside for the people of Auckland in 1845, with planting commencing in 1850 (Gill 1989; Wilcox *et al.* 2004). The park supports the largest contiguous patch of forest habitat within the city's centre with over 70 ha of exotic and native forest, much planted in the mid to late 1800s, but with some remnant examples of native canopy species in the gullies. Examples of exotic canopy trees include oak (*Quercus robur*), cottonwood (*Populus deltoides*), and large araucaria trees including Norfolk Island pine (*Araucaria heterophylla*), Cook pine (*A. columnaris*), and bunya (*A. bidwillii*). Native canopy species includes karaka (*Corynocarpus laevigatus*), pūriri (*Vitex lucens*), kauri (*Agathis australis*), rimu (*Dacrydium cupressinum*), tōtara (*Podocarpus totara*), and tanekaha (*Phyllocladus trichomanoides*). Canopy heights within this forested area can reach to >20 m, with a regenerated understory of predominantly native species. This large mixed forest provides an opportunity to study the structure of an urban bird community.

Five-minute bird counts (hereafter 5MBC) are a technique widely used in New Zealand to describe composition and change in forest bird populations (Dawson & Bull 1975; Hartley 2012). In 1987 and 1988 Gill (1989) used the 5MBC technique to report on the structure of the bird community within the domain's forest habitat. Here, we report the results of a similar survey conducted 30 years later to explore potential changes in the structure of this community in the heart of New Zealand's largest city.

METHODS

We conducted 5MBCs at two sites established by Gill (1989) being 600 m apart and surrounded by forest for 100 m in all directions. At these sites we made c. 15 counts per month, typically across four to six count days, from April 2019 to April 2020 (total counts 1987–88 = 195; 2019–20 = 184). Count days were typically spaced across a month as best as possible allowing for suitable weather conditions. 5MBC methodology followed Dawson & Bull (1975), with observers recording all birds seen and or heard whilst stationary at each count site. On a given count day three to four counts were made by experienced observers between the two counting sites, with either a single observer doubling back and forth between sites or two observers swapping between sites. Repeat counts at the same site on a day were made at least 25 min apart. Counts were made between the hours of 0900 and 1500, and only conducted in fine to reasonable weather, lacking rain or strong winds. In total five experienced observers conducted the counting work.

We utilised both multivariate parametric and nonparametric methods to assess changes in the avifauna assemblage between survey years. We used a Generalised Linear Mixed Model (GLMM) approach (Bolker *et al.* 2009) to investigate the differences in species richness (overall, native, and introduced), overall abundance, and the abundance of individual species between survey years. Models were fitted in R 4.2.3 (R Core Team 2023) using the glmmTMB function in the *glmmTMB* package. For all models survey year (1987–1988 or 2019–2020) was included as a fixed effect, along with season (spring, summer, autumn, winter) and time of day (minutes after sunrise) to account for expected seasonal and diurnal variation in counts. Site ID was included as a random effect, to account

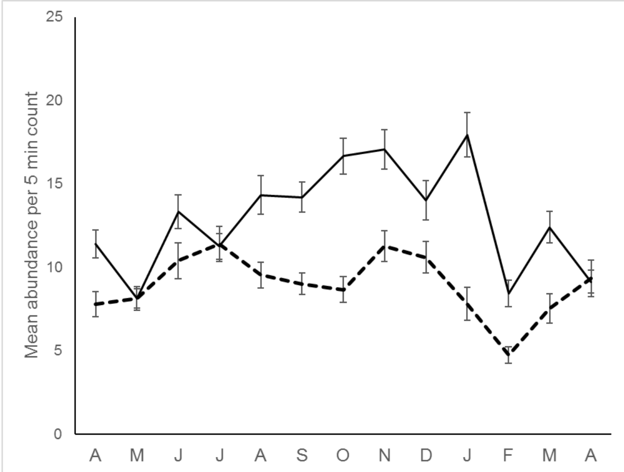

Figure 1. Monthly mean total abundance from 5-min bird counts at two sites in the Auckland Domain forest in 1987–88 (hashed line) and 2019–20 (solid line). Error bars represent ± 1 standard deviation.

Table 2. Summary of generalised linear mixed model (GLMM) results testing for an effect of survey year on community structure measures and individual species abundances in the Auckland Domain Forest, Auckland, New Zealand. Lines shaded grey indicate models where survey year had a significant effect on the response variable.

		Survey year (Reference level: 1987/88)	Season‡ (Reference level: Winter)				Time of day (min after sunrise)‡	Max. daily temp (°C)‡#	Min. daily temp (°C)‡#	Mean daily wind speed (km h-1)‡#	Daily rainfall (mm)‡#
	Distribution†	2019/20	Spring	Summer	Autumn	LRT					
Overall community structure responses											
Overall species richness	N	1.788***	1.308	0.726	-0.249	***	-0.002***	-0.094*	-0.020	-0.033***	0.011
Native species richness	N	0.650***	0.675	0.396	0.056	***	-0.001	-0.056*	0.003	-0.018**	-0.007
Introduced species richness	N	1.139***	0.635	0.332	-0.304	***	-0.001**	-0.038	-0.024	-0.015*	0.018
Overall abundance	NB2 (25.5)	0.457***	0.227	0.125	-0.107	***	-0.001**	-0.015	-0.009	-0.004	-0.003
Individual responses: Native species											
Silvereye	NB1 (1.05)	-0.329***	-0.286	0.056	-0.034		-0.0004	-0.016	-0.017	0.002	-0.017
Tūī	P	2.413***	0.235	0.190	-0.450	***	0.0001	-0.009	0.041	-0.006	-0.016
New Zealand fantail	P	-0.057	0.126	-0.208	0.174	*	-0.0002	-0.001	-0.005	-0.006	0.001
Grey warbler	P	0.770***	0.620	0.101	-0.148	***	-0.001	-0.049	-0.013	-0.009	-0.006
Sacred kingfisher	NB1 (0.16)	-0.103	2.187	1.084	-0.361	***	-0.002	0.053	-0.013	-0.018	-0.006
Kererū	NB1 (0.44)	2.016*	-18.77	1.167	0.745		-0.001	-0.080	0.133	-0.104	-0.032
Individual responses: Introduced species											
Eurasian blackbird	P	0.318***	0.441	0.054	-0.078	***	-0.0002	0.026	-0.040*	0.003	0.001
Eastern rosella	ZIP	4.727***	0.460	-0.267	0.144		-0.001	-0.030	-0.017	-0.0003	-0.002
Chaffinch	NB2 (1.67)	0.074	1.132	1.605	0.465	***	-0.003**	0.009	-0.093*	0.008	0.006
Song thrush	ZIP	0.175	0.308	0.170	-1.740	***	-0.0000	-0.034	0.020	-0.029*	0.024
House sparrow	NB1 (0.82)	2.284***	-0.089	0.705	-0.954	**	-0.001	-0.127	0.031	-0.026	-0.014
European goldfinch	ZIP	1.434***	1.586	0.821	0.445	***	0.002	-0.120	-0.048	-0.015	0.031
European greenfinch	NB2 (0.78)	-0.716*	-0.459	-0.820	-1.273		-0.003*	0.009	-0.056	-0.009	-0.114*
Common myna	NB2 (0.57)	2.357***	-2.166	-0.176	0.522	*	-0.002	0.123	-0.068	0.002	0.006
European starling	NB2 (0.44)	0.013	0.851	1.212	0.860		-0.001	-0.623***	0.200	-0.033	0.040

Parameter estimates are presented for each model term at the reference levels stated. Whole effects were tested with likelihood ratio tests (LRTs); significant chi-square test statistics from LRTs are indicated with: *, p <0.05; **, p <0.01; and ***, p <0.001. Species are listed by mean abundance across all counts (highest first). † Error distribution used for model: N = normal, NB = negative binomial (dispersion parameter estimate given in parentheses), P = Poisson, ZIP = zero-inflated Poisson. ‡ Included in the models as control variables. #Data from NIWA National Climate Database.

for correlation of repeated measures at the same sites. In addition, four meteorological variables were also included in the models as controls for changing climate over the timeframe: maximum daily temperature (°C), minimum daily temperature (°C), mean daily wind speed (km h^{-1}), and daily rainfall (mm). These metadata were sourced from NIWA National Climate Database (Auckland Aero weather station; https://cliflo.niwa.co.nz/, accessed 29 Jul 2020). Each count response variable was modelled using the best-fitting distribution as determined with the *fitdistrplus* package (Delignette-Muller & Dutang 2015) in R (Table 1).

To analyse changes in composition of the avian community between survey years, we used non-metric multidimensional scaling (NMDS; Kruskal 1964), permutational multivariate analysis of variance (PERMANOVA; Anderson 2001), and permutational analysis of multivariate dispersions (PERMDISP; Anderson 2006). Rare species (those present in <2% of counts; Table S1) were removed for all analyses (McCune & Grace 2002). As we were interested in changes involving dominant species within bird communities, no transformation was applied to the data before construction of the distance-matrix. NMDS ordinations were performed to visualise differences in bird assemblages, using the metaMDS function of the *vegan* package (Oksanen *et al.* 2013) in R 4.2.3. Species centroids were plotted separately to aid interpretation of observed differences in community structure. PERMANOVA was used to test whether community composition varied between survey years (Anderson & Walsh 2013), performed using the adonis2 function of *vegan* (999 permutations). A PERMDISP analysis was used to test for a difference in the variability of bird assemblages between survey years (Anderson & Walsh 2013), using the betadisper and permutest functions of *vegan* (constraining permutations within sites; based on 999 permutations).

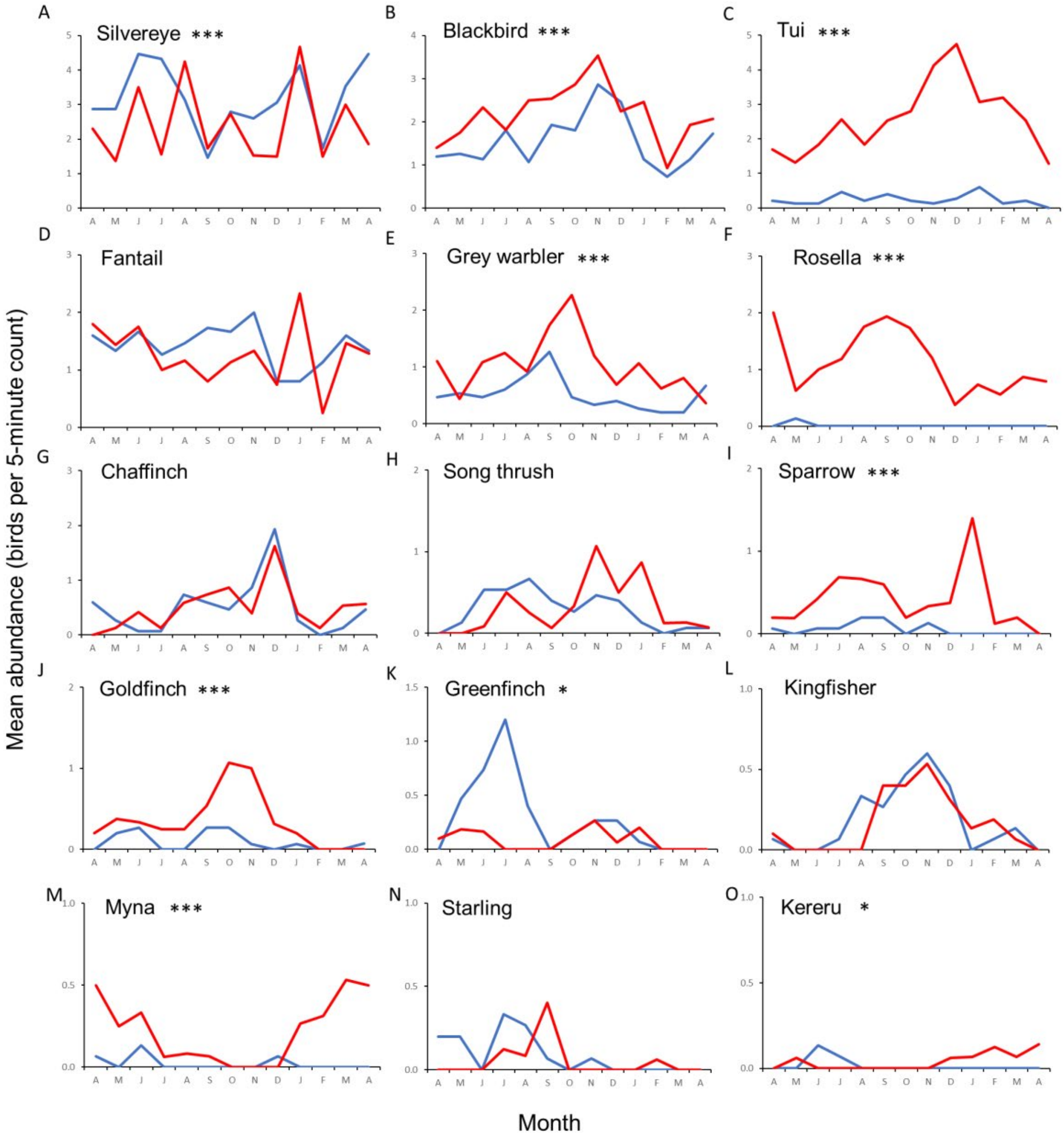

Figure 2. Mean abundance of the 15 most commonly occurring bird species present in Auckland Domain forest during year-long 5-min bird count surveys in 1987–88 (blue line) and 2019–20 (red line). Species are listed in order of mean abundance across all counts (N = 379). Note: y-axis scale varies with species. Asterisks represents statistical significance based on results of GLMM testing (see Table 2).

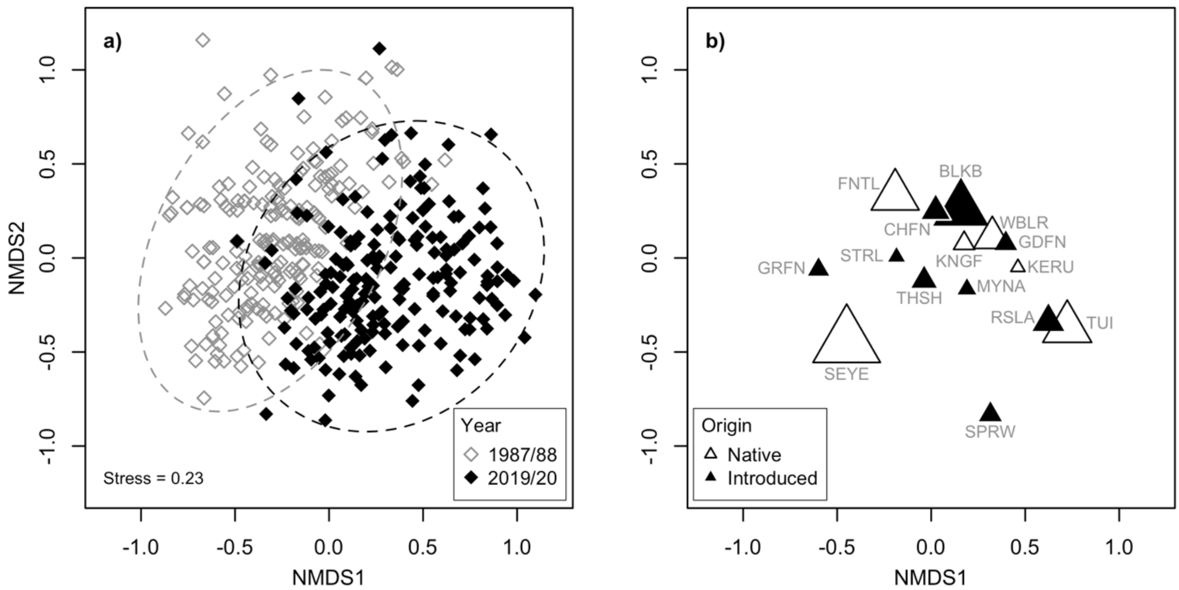

Figure 3. (a) NMDS ordination of avian community composition in 1987–88 vs. 2019–20 with hashed ellipses outlining the 95% confidence intervals for each survey year. (b) Species centroids show the relationships among species as defined by their relative abundance during both survey periods (points scaled by proportion of total abundance). The stress value (0.23) indicates the degree of distortion between the multidimensional data and its two-dimensional representation, with values below 0.2 considered good and values between 0.2 and 0.3 providing an acceptable but moderate fit. Species abbreviations: BLKB, Eurasian blackbird; CHFN, chaffinch; FNTL, fantail; GDFN, goldfinch; KERU, kererū; KNGF, sacred kingfisher; MYNA, common myna; RSLA, eastern rosella; SEYE, silvereye; SPRW, house sparrow; THSH, song thrush; TUI, tūī; WBLR, grey warbler.

Table 3. Summary of PERMANOVA results for the effect of survey year (1987–88 vs. 2019–20) on avian community structure in the Auckland Domain forest. *F*-values (pseudo-*F*) are derived from 999 permutations.

Factor	d.f.	F	R^2	P
Survey year	1	85.0	0.172	0.001***
Season	3	8.94	0.054	0.001***
Site ID	1	8.17	0.017	0.001***
Residuals	373		0.757	
Total	378			

RESULTS

A total of 20 bird species was recorded in the domain forest over both 1987–88 and 2019–20 survey periods, eight of which were native species and 12 introduced (Table 1). Seventeen species were recorded in 1987–88, increasing to 20 in 2019–20. Three additional species (not included in analyses) were recorded flying over during 2019–20 counts: red-billed gull | tarāpunga *Chroicocephalus novaehollandiae*, southern black-backed gull | karoro *Larus dominicanus*, and paradise shelduck | pūtangitangi *Tadorna variegata*. Tūī *Prosthemadera novaeseelandiae*, Eurasian blackbird *Turdus merula*, and silvereye | tauhou *Zosterops lateralis* were the three most frequently encountered bird species in 2019–20 surveys, present in 90.2%, 88.0% and 71.7% of surveys, respectively (Table S1). In comparison, the three most observed species in 1987–88 were silvereye (91.8% of surveys), New Zealand fantail | pīwakawaka *Rhipidura fuliginosa*, 83.1%), and blackbird (78.5%). Similarly, the most abundant species in 2019–20 counts was tūī (mean ± SE = 2.7 ± 0.12 birds/5-min count), whereas in 1987–88 counts it was silvereye (3.2 ± 0.18 birds/5-min count; Table 1).

Survey year had a significant effect on overall community structure measures, even after accounting for season, time of day, temperature, wind, and rain (GLMMs, Table 1), with overall species richness, native species richness, introduced species richness and overall abundance all higher in 2019–20 compared to 1987–88 (Fig. 1). Three native species (tūī, grey warbler | riroriro *Gerygone igata*, and kererū | New Zealand pigeon *Hemiphaga novaeseelandiae*) showed an increase in abundance in 2019–20, while one native species decreased (silvereye; Table 1, Fig. 2). The abundance of five introduced bird species also showed a significant increase in 2019–20 (blackbird, eastern rosella *Platycercus eximius*, house sparrow *Passer domesticus*, goldfinch *Carduelis carduelis*, and common myna *Acridotheres tristis*). European greenfinch *Chloris chloris* declined in abundance between survey years (Table 1, Fig. 2).

The NMDS ordination plot provided further evidence of a divergence in avian communities between 1987–88 and 2019–20 survey periods, supported by PERMANOVA analyses (Table 2), with the group centroid of 2019–20 counts shifting significantly to the right (Fig. 3). Survey year explained a greater amount of variation in community composition (R^2 = 0.17) in comparison to season or site variation (R^2 = 0.05 and 0.01, respectively; Table 3). However, PERMDISP analyses indicated that variability in community composition was not significantly different between years (PERMDISP; F = 2.02, df = 1, P = 0.16).

DISCUSSION

Increases in overall abundance and species richness of native and introduced birds in the domain forest is encouraging and shows the importance of mature urban forest habitats for supporting healthy urban avian communities. Though species specific detection probabilities have not been accounted for in our analysis, changes between counts of Gill (1989) and our current survey are most likely driven by a combination of forest habitat succession and increasing levels of pest control. In the three decades since previous counts, forest growth as well as pest plant control by Auckland Council of species such as tree privet (*Ligustrum lucidum*) has resulted in a reduction of competition for native shrubs and ground cover plant species, and healthy succession of diverse canopy, subcanopy and understory habitats providing food supply and nesting habitats for a range of native and introduced birds (Boffa Miskell 1993; Wilcox *et al.* 2004). In addition, in 1980s there was no pest control within the domain (B.J. Gill *pers. comm.*) in contrast to the sporadic trapping and poisoning of rodents and possums by contractors and local conservation volunteers at present under the Auckland Council Regional Pest Management Strategy (Auckland Council 2020). Control of pests in urban forests can benefit the bird community by either release of direct predation or reduction food competition from forest browsers such as possums. For example, in Wellington populations of native and introduced birds increased in the 1990s following regional pest control which also resulted in rarer endemics such as bellbird|korimako *Anthornis melanura*, kākāriki|red-crowned parakeet *Cyanoramphus novaezelandiae* and kākā *Nestor meridionalis* recolonising the city several years before they were translocated to the region following the creation of the pest-free Karori Wildlife Sanctuary (Miskelly *et al.* 2005; Brockie & Duncan 2012; Miskelly 2018).

The changing climate over recent decades could also have a contributory role in the observed bird assemblage differences between survey years. The effect of climate change on the domain bird assemblage cannot be fully understood here, without a multi-year dataset to disentangle long term trends from short term seasonal and natural interannual variation. However, by including daily weather variables in the models as a proxy, we can, at a minimum, discern which species counts are significantly affected by these, and control for gross differences in temperature, wind, and rainfall between survey years. Furthermore, the effect size of the weather variables on count numbers was, for most species, observed to be much smaller than the effect size for survey year, supporting the argument that factors other than weather were the major driver of observed community change.

The restoration of forest habitats can have complex and species-specific effects of forest avifauna composition (Fea *et al*. 2020; Binny *et al*. 2020). The approximately 30 years between year-long counts in the domain's forest saw an increase in abundance of forest-adapted endemic species including tūī, kererū, and grey warbler, and a decline in generalist natives silvereye and fantail. The NMDS structure analysis demonstrates the difference between these communities, which were structured around a dominance of silvereye and fantail in the 1980s and tūī in the 2020s (Fig. 2). These data are consistent with other studies of mainland forest avifaunas that have received conservation management. For example, Miskelly (2018) showed that where habitats are restored through invasive mammal removal and numbers of endemic forest taxa increased through targeted translocations, forest adapted endemic species can outcompete more common and widespread taxa that tend to be habitat generalists. However, these community changes appear to be context-specific, with Spurr & Anderson (2004) reporting an increase in both forest-adapted tūī and generalist grey-warbler following the eradication of possums from Rangitoto Island. Interestingly, our data also indicate an increase in a number of introduced bird species between our count years in the Auckland Domain; it may be that in urban forests, with more limited number of endemic species and a lack of complete pest removal, that the suppressing effects of endemic habitat dominance is reduced.

The large increase of tūī in the domain forest between counts in 1987–88 and 2019–20 is noteworthy. This result is consistent with other studies showing increases in this iconic endemic bird across the Auckland isthmus over the past 30 years (Spurr & Anderson 2004; Lovegrove & Parker 2023). Tūī are mobile and move seasonally between islands in the Hauraki Gulf, larger rural forests on the city's periphery and Auckland urban forests (Stewart & Craig 1985). Recent research has established that pest mammal control or eradication can lead to landscape scale spillover over of tūī into surrounding habitats (Fitzgerald *et al.* 2019). It is likely that pest eradication and/or revegetation projects on Hauraki Gulf islands near to Auckland (e.g., Rangitoto, Motutapu, Tiritiri Matangi, The Noises, and Rotoroa Island), and landscape-scale pest control programs in the large, forested parks such as the Hunua and Waitakere Ranges on the city's fringe is significantly benefitting tūī numbers across the region, including in urban forests (Lovegrove & Parker 2023).

Introduced bird species, including Eurasian blackbird, eastern rosella, house sparrow, European goldfinch, and common myna, also increased in abundance in the domain forest between 1987–88 and 2019–20. As with native taxa, these species have likely benefitted from habitat maturation and control of mammalian predators and browsers. However, for two of these species, regional colonisation histories and cavity-nesting behaviour may play role in explaining increases. Eastern rosella were established by introductions in Auckland prior to 1920, and by 1960 had moved into Northland with a slower progression into the Waikato (Fleming 1944; Wright & Clout 2001). Rosellas had an incomplete distribution across Auckland between 1969 and 1979 (Bull *et. al.* 1985); however, they were present in all areas of the region in 1999–2004 and had expanded southwards into the central North Island (Robertson *et. al.* 2007). The single occurrence of this species in counts by Gill (1989) is consistent with these observations, suggesting that by the 1980s the species was near to occupying the entire region albeit at lower abundance (Wright & Clout 2001). Rosellas had increased significantly within Auckland by 2020, being the fifth most common species in counts in the domain forest. Common myna were introduced to the South Island and the west and east coasts of the central and southern North Island between 1869 and 1883 (Beesley *et al.* 2023). In the North Island the species reached Auckland in the late 1940s and early 1950s (Cunningham 1948, 1951 & 1954; Beesley *et al.* 2023). Our data suggest increasing numbers of birds between 1987–88 and 2019–20, particularly during the summer breeding season, which is also consistent with observed increases of the species in six Northland forests across a similar time period (Pierce *et al.* 1993). Additionally, as a cavity-nesting species, the maturation of forest is likely to have a positive effect on eastern rosella and common myna numbers by increasing availability of nest sites as natural cavities form over time in older trees (Galbraith *et al.* 2014). This is concerning, as both species directly compete with native bird species for nest-cavity resources, which are already typically limited in young forest particularly in systems where cavity-excavating species (e.g., woodpeckers, Picidae) are absent (Galbraith *et al.* 2014; Krull *et al.* 2015).

In conclusion, comparison of birds counts within Auckland Domain between 1987–88 and 2019–20 show an increase in the overall abundance and species richness of native and introduced birds. Of note was the increase in forest-adapted endemic species and a decline in generalist natives, which has also been observed in other studies where forest succession is aided by management actions such native replanting and pest plant and mammal control at a local scale. However, external factors can also drive changes in urban avian communities. The large increase in numbers of highly mobile tūī visiting the domain forest has likely been driven by conservation management actions at a regional level, whereas changes in abundance of introduced eastern rosella and common myna reflect the timeline of their respective invasion histories. Overall, our results reinforce the need to protect maturing urban forest habitats, supporting the native forest recovery within these forest matrices, particularly to enhance native bird populations, whilst thinking about recovery of urban bird populations at a landscape scale.

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