Original Research Article

Title:

Genomic sequencing confirms absence of introgression despite past hybridisation between a

common and a critically endangered bird

Running head:

Genomic data finds no introgression in kakī

Authors:

Natalie J. Forsdick[1], Denise Martini[1], Liz Brown[2], Hugh Cross[1], Richard F. Maloney[3],

Tammy E. Steeves[4], Michael Knapp[1]

1.   Department of Anatomy, University of Otago.

2.   New Zealand Department of Conservation, Twizel, New Zealand.

3.   New Zealand Department of Conservation, Christchurch, New Zealand.

4.   School of Biological Sciences, University of Canterbury, Christchurch, New Zealand.

Corresponding author:

Natalie J. Forsdick

Email: natalieforsdick@gmail.com

Present address: 156A Gowing Drive, Meadowbank, Auckland 1072, New Zealand

Abstract

Genetic swamping resulting from interspecific hybridisation can increase extinction risk for threatened species. The development of high-throughput and reduced-representation genomic sequencing and analyses to generate large numbers of high resolution genomic markers has the potential to reveal introgression previously undetected using small numbers of genetic markers. However, few studies to date have implemented genomic tools to assess the extent of interspecific hybridisation in threatened species. Here we investigate the utility of genome-wide single nucleotide polymorphisms (SNPs) to detect introgression resulting from past interspecific hybridisation in one of the world's rarest birds. Anthropogenic impacts have resulted in hybridisation and subsequent backcrossing of the critically endangered Aotearoa New Zealand endemic kakī (black stilts; *Himantopus novaezelandiae*) with the non-threatened self-introduced congeneric poaka (Aotearoa New Zealand population of pied stilts, *Himantopus himantopus leucocephalus*), yet genetic analyses with a limited set of microsatellite markers revealed no evidence of introgression of poaka genetic material in kakī, excluding one individual. We use genomic data for ~63% of the wild adult kakī population to reassess the extent of introgression resulting from hybridisation between kakī and poaka. Consistent with previous genetic analyses, we detected no introgression from poaka into kakī. These collective results indicate that, for kakī, existing microsatellite markers provide a robust, cost-effective approach to detect cryptic hybrids. Further, for well-differentiated species, the use of genomic markers may not be required to detect admixed individuals.

1 Introduction

Growing evidence from genetic and genomic data indicates that interspecific hybridisation (hereafter, hybridisation) has been integral in the evolutionary history of many species (Mallet, 2008), challenging existing perceptions of the intrinsic value of hybrids and hybrid species, and further highlighting the complexity of conservation policy relating to them (Haig & Allendorf, 2006; Jackiw et al., 2015; Wayne & Shaffer, 2016). Hybridisation may improve individual fitness, population resilience and adaptive potential (Arnold, 1997; Dowling & Secor, 1997; Mallet, 2007; Seehausen, 2004), and as such hybridisation between closely related species or subspecies has been proposed as a conservation management tool to assist genetically depauperate threatened species by introducing novel genetic variation to increase genetic diversity and improve fitness (Arnold, 2016; Harrisson et al., 2016; Ingvarsson, 2001; 65 Mallet, 2005). Nevertheless, potential tradeoffs resulting from conservation management of hybrids and hybridisation – especially recent human-induced hybridisation between threatened endemic and non-threatened non-endemic congeners – warrants careful ethical and practical consideration, balancing conservation priorities alongside the ecological, social, and economic costs-benefits of hybrids (e.g., Hamilton & Miller, 2016; Estévez et al., 2015; Schlaepfer et al., 2011).

Conservation is an inherently values-based discipline. From a conservation perspective, threatened endemic species are valued over non-threatened non-endemic species due to their rarity and their known (or perceived) ecological importance, which generally leads to ethical and moral obligations to conserve them, especially if they are also culturally significant species (Booth et al., 2011; Courchamp et al., 2006; Maguire & Justus, 2008; Richardson &

Loomis, 2009). However, the potential conservation value of hybrids should not be viewed as

static, especially as altered species ranges increase the prevalence of hybrids (Chunco, 2014),

and the inclusion of genomic data continues to improve our understanding of the impacts of

hybridisation (vonHoldt et al., 2018). Further, there are growing calls for researchers identify

and avoid unscientific and biased perspectives on hybrids and hybridisation in conservation

(Draper et al., 2021; Hirashiki et al., 2021).

Regardless of the perceived value of hybrids, hybridisation negatively impacts threatened

species recovery through the misplaced reproductive efforts of interspecific breeding

(Allendorf et al., 2001). This can reduce the reproductive outputs of species of conservation

concern by demographic swamping (Allendorf et al., 2001; Wolf et al., 2001). Another

potential impact of hybridisation is when subsequent backcrossing to the parental species

incorporates genetic material from one species into the genome of another, known as

introgression (Rhymer & Simberloff, 1996). Negative impacts of introgression may include

outbreeding depression, where the breakdown of coadapted gene complexes or the

introduction of maladaptive traits results in the decreased fitness of hybrid offspring (Arnold,

1997; Edmands, 2007; Lynch, 1991), and, at its most extreme, may result in extinction-by-

hybridisation (Allendorf et al., 2001; Fitzpatrick et al., 2010; Quilodrán et al., 2018; Rhymer

& Simberloff, 1996; Riley et al., 2003; Taylor et al., 2006; Todesco et al., 2016).

Genetic tools may be employed to assist conservation management programmes in assessing

the extent and impacts of hybridisation, and for identification of cryptic hybrid offspring

morphologically indistinguishable from parental types (Chan et al., 2006; Ma & Lambert,

1997; Milián-García et al., 2015; Pierpaoli et al., 2003). However, to date, most conservation-

relevant studies have used a small number of genetic markers (e.g., microsatellites) that may

not be representative of genome-wide diversity, particularly among threatened species where

population bottlenecks have left populations genetically depauperate (Taylor, 2015; Taylor et

al., 2015; Väli et al., 2008). Over the past decade, genomic sequencing technologies have

progressed, and rapidly declining costs now enable the sequencing and assembly of complete

genomes for threatened non-model organisms (e.g., Li et al., 2010; Sutton et al., 2018), or the

generation of thousands of genomic markers (i.e., single-nucleotide polymorphisms (SNPs))

distributed throughout the genome via reduced-representation sequencing, sufficient to

facilitate population-level estimation of metrics including diversity, relatedness, population

structure, and introgression in an efficient, cost-effective manner (Ba et al., 2017; Chen et al.,

2016; Peek et al., 2019; Rexer-Huber et al., 2019; Rick et al., 2019). Population-level

reduced-representation sequencing (including restriction-enzyme associated DNA sequencing

(RADseq; Baird et al., 2008), double-digest RADseq (ddRADseq; Peterson et al., 2012), and

genotyping-by-sequencing (GBS; Elshire et al., 2011)) is an approach that can produce

thousands of variant sites for high-resolution population genomic analyses (Davey et al.,

2011; Davey & Blaxter, 2010; Narum et al., 2013) and as such has wide applicability for

conservation (Andrews et al., 2016; Seabury et al., 2011; Wright et al., 2019). Large genomic

marker sets are expected to provide greater power to resolve questions relating to

hybridisation and introgression (similar to that observed when estimating genetic diversity

and differentiation (Fischer et al., 2017) and relatedness (Galla et al., 2020)). However,

studies comparing the enhanced resolution of genomic data with previous genetic analyses

related to hybridisation in a conservation context are thus far limited (Table 1). Here we

demonstrate the utility of genomic tools for assessment of the impacts of hybridisation in a

critically endangered wading bird.

Table 1 Interspecific hybridisation research in a conservation context where both genetic and genomic data have been generated and both have
been used to assess hybrids or hybridisation. 'Conservation relevance' is used to mean that the hybridisation involves a threatened species, and
the outcomes of the study have implications for conservation and/or present recommendations for conservation management. Web of Science
search (19 July 2020): ALL FIELDS: (((genetic AND genomic) AND (hybridization OR introgression OR admixture) AND (conservation OR
threatened OR endangered)) AND (fish OR bird OR mammal OR marsupial OR amphibian OR reptile OR plant OR invertebrate OR insect OR
mollusc OR crustacean)). Of the 237 results, to capture empirical research that used high-throughput genomic sequencing techniques only,
reviews and editorials, and articles published prior to 2014 were excluded. From the remaining 119 results, studies assessing intraspecific
admixture or with no clear conservation relevance were excluded.

Taxa	Hybridisation relevance	Genetic data	Genomic data	Genetic and genomic data comparison	Conservation implications
Mottled duck (*Anas fulvigula*) and mallard (*A. platyrhynchos*)[1]	Species known to hybridise with subsequent backcrossing.	Microsatellite markers, introns, allozymes	ddRADseq	Genomic data provides more robust evidence of admixture with mallards.	Hybridisation creates potential for replacement of mottled ducks by mallards.
Golden-winged warbler (*Vermivora chrysoptera*) and blue-winged warbler (*V. cyanoptera*)[2]	Species morphologically differentiated, but genetically indistinguishable using genetic markers.	Allozymes, AFLPs, introns, microsatellite markers, mitochondrial markers	ddRADseq, whole-genome resequencing	Species indistinguishable based on genetic and ddRADseq data. Whole-genome resequencing data differentiates species and identifies both historical natural gene flow and recent human-mediated gene flow.	Taxonomic recommendation to retain species designations, and to use both genetic and phenotypic data to inform conservation.
Spotted owl (*Strix occidentalis*) and barred owl (*Strix varia*)[3]	Species known to hybridise with subsequent backcrossing.	Microsatellite markers, AFLPs	Whole-genome resequencing	Genetic data limited to identifying F1 and F2 hybrids. Genomic data provides no evidence of widespread introgression.	Hybridisation creates potential for replacement of the endangered spotted owl by the barred owl.
Przewalski's horse (*Equus ferus przewalskii*), and domesticated horse (*E. caballus*)[4]	Species known to hybridise with subsequent backcrossing.	Mitochondrial markers, sex chromosome markers	Whole-genome resequencing	Genomic data provides more nuanced evidence of both historical natural gene flow and recent human-mediated gene flow in captivity.	Genomic data will be used to limit admixture in the Przewalski's horse conservation breeding program.
Eastern massasauga rattlesnake	Western edge of Eastern massasauga range	Nuclear markers,	RADseq	Limited population structure identified using genetic data.	Recommendation to categorise the admixed population as a 'Distinct

(*Sistrurus catenatus*) and Western massasauga rattlesnake (*S. tergeminus*)[5]	displays evidence of historic admixture with nearby non-threatened Western massasauga.	mitochondrial markers		Genomic data provides evidence of additional population structure, including admixture attributed to historical natural gene flow.	Population Segment' under the Endangered Species Act (USA) due to its 'discreteness, significance, and population status'.
European pool frog (*Pelophylax lessonae*), edible frog (*P. esculentus*), Italian pool frog (*P. bergeri*)[6]	Species complex readily hybridises, including invasive *P. bergeri* with native *P. lessonae* and the hybridogenetic *P. esculentus*.	Mitochondrial markers, introns, microsatellite markers	RADseq	Genetic data identified admixed lineages but could not distinguish between hybridisation or incomplete lineage sorting. Genomic data confirms species complex as a hybrid swarm across most of the range, excluding one population of *P. lessonae*.	Genetically distinct population of *P. lessonae* may warrant conservation priotisation.
Devils Hole pupfish (*Cyprinodon diabolis*) and other *Cyprinodon* spp.[7]	Many pupfish species readily hybridise and produce viable offspring.	Mitochondrial markers, microsatellite markers	ddRADseq	Genomic data detects recent gene flow between *C. diabolis* and both *C. amargosae* and *C. pectoralis*.	Hybridisation creates potential for replacement of the Devils Hole pupfish, as has been common in the evolutionary history of the genus.
Montipora spp. (corals)[8]	Taxonomic uncertainty, including potential introgressive hybridisation.	Mitochondrial markers, nuclear markers	RADseq	Genetic data clarified taxonomy, but could not distinguish introgression, incomplete lineage sorting, or phenotypic plasticity. Genomic data further resolves taxonomy and confirms introgression, but timing difficult to resolve due to recent origin of the species complex.	Revised taxonomy will inform conservation.
Nautilus spp. (molluscs)[9]	Taxonomic uncertainty, including potential introgressive hybridisation.	Mitochondrial markers, microsatellite markers	ddRADseq	Genetic data insufficient for taxonomic delimitation. Genomic data resolves taxonomy, including evidence of recent admixture between species.	Revised taxonomy will inform conservation.

1. Peters et al., 2016; 2. Toews et al., 2016; 3. Hanna et al., 2018; 4. Der Sarkissian et al., 2015; 5. Sovic et al., 2016; 6. Dufresnes & Dubey, 2020;

7. Martin et al., 2016; 8. Cunha et al., 2019; 9. Combosch et al., 2017.

One of the world's rarest bird species, the Aotearoa New Zealand kakī (black stilt,

Himantopus novaezelandiae) provides a quintessential example of a threatened species

affected by human-induced hybridisation (BirdLife International, 2018; Robertson et al.,

2016). Anthropogenic impacts resulted in population decline during the 1900s, with numbers

falling to approximately 23 individuals comprising a single population in Te Manahuna/the

Mackenzie Basin in 1981 (Pierce, 1984b; Steeves et al., 2010). Adaptive conservation

management of kakī, including predator control throughout Te Manahuna and a programme

of captive breeding and rearing for translocation have been integral to increase the kakī

population to 169 wild adults in 2020 (Hagen et al., 2011; Heezik et al., 2005; Keedwell et

al., 2002; Maloney & Murray, 2001; Reed et al., 1993; Steeves et al., 2010).

Research to date based on pedigree and genomic data indicates that despite the population

history of kakī, the population exhibits relatively low levels of inbreeding and relatedness

among individuals (Galla, 2019; Galla et al., 2019). However, distorted allele frequencies in

small populations can produce an apparent excess of heterozygotes, which would also result

in underestimated levels of inbreeding, and such skewed results can be difficult to correctly

assess in the absence of a comparable large, outbred population (Kardos et al., 2015).

Inbreeding depression in kakī manifests reduced hatching success (Hagen et al., 2011), and so

breeding has been carefully managed by the DOC's Kakī Recovery Programme (summarised

by Galla et al., 2020). Along with predation and altered habitat availability, interspecific

hybridisation may pose a threat to species recovery (Steeves et al., 2010). The Aotearoa New

Zealand population of congeneric Australian pied stilts (hereafter referred to as poaka;

Himantopus himantopus leucocephalus) self-introduced from Australia at least 200 years

ago, and anthropogenic impacts facilitated expansion of the species' range (Pierce, 1984b).

Limited mate choice and a male sex bias among kakī when kakī numbers were declining

promoted hybridisation between kakī and poaka, producing viable, fertile hybrid offspring

(Pierce, 1984a; Steeves et al., 2010).

Current kakī conservation management policies reflect the conservation value of non-

admixed kakī, which are individuals with pure-black plumage that genetically assign to kakī

based on a small set of mitochondrial and microsatellite markers (Maloney & Murray, 2001;

Reed et al., 1993; Steeves et al., 2010). Steeves et al. (2010) confirmed the genetic

distinctiveness of kakī, and found no evidence of introgression from poaka except in a single

individual. This finding was attributed to outbreeding depression that negatively impacts

fitness of female hybrid offspring, combined with an ephemeral skewed sex ratio and active

management to exclude hybrids (Steeves et al., 2010). However, Steeves et al. (2010)

acknowledged that the small marker set used may not be representative of genome-wide

variation, and thus may lack the power to detect low levels of introgression resulting from

hybridisation (see also Brumfield, 2010).

Here, we use population-level GBS and a reference-guided approach to SNP discovery for

kakī, known hybrids, poaka, and Australian pied stilts. We infer the genomic extent and

pattern of introgression due to hybridisation between kakī and poaka in the contemporary

kakī population, as compared with that of previous genetic analyses. Determining the utility

of genetic and genomic markers for detection of introgressive hybridisation is essential not

only for the conservation of kakī, but also for detecting hybridisation in threatened species

more broadly.

2 Materials and Methods

2.1 Sample collection and DNA extraction

Following Steeves et al. (2010), individuals sampled herein were grouped by plumage phenotype ('node') as categorised by Pierce (1984a) and used by the Department of Conservation's Kakī Recovery Programme. Poaka and pied stilts (plumage nodes A–C2; Pierce, 1984a) were labelled 'pied', completely black node J individuals were labelled 'kakī', and individuals of intermediate plumage (nodes D1–I/J) or known hybrid parentage were labelled 'hybrid' (Supplementary Table 1). Extracted genomic DNA (gDNA) was available from 80 stilt samples including kakī, Australian pied stilts and poaka, and hybrids from previous genetic analyses.

We extracted gDNA from an additional 155 feather samples collected as part of regular handling practices for kakī under Aotearoa New Zealand's Department of Conservation (DOC) ethics approvals (AEC #283) at the DOC's Kakī Recovery Programme, Twizel, and the Isaac Conservation and Wildlife Trust kakī captive rearing facility, Christchurch, Aotearoa New Zealand. Pedigree information is recorded for all kakī individuals as part of routine Kakī Recovery Programme management, extending up to seven generations. In addition, blood samples were collected as part of routine health checks from two Australian pied stilts at Adelaide Zoo (provided under a Royal Zoological Society of South Australia Specimen Licence Agreement; Import Permit #2016061954), two node B poaka from Auckland Zoo (under Auckland Zoo animal ethics approval), along with a tissue sample from one poaka from Hawke's Bay, North Island (Figure 1). Poaka from the North Island were preferentially sampled due to a low likelihood of recent contact with kakī, minimising the chance of these individuals having recent hybrid ancestry. We extracted gDNA from all samples using a Thermo Scientific™ MagJET™ Genomic DNA kit, following Protocol E

(manual genomic DNA purification from up to 20 mg tissue). We quantified gDNA for all

samples using a NanoDrop™ 8000 Spectrophotometer to assess viability for GBS. GBS

guidelines from our sequencing provider (AgResearch Ltd., Mosgiel, Aotearoa New Zealand)

recommended a total of 1 µg of DNA per sample at a concentration of 80–150 ng/µL, and

260/280 and 260/230 ratios of 1.80–2.00 and > 1.0 respectively.

Figure 1 Locations of kakī captive breeding facilities and wild breeding distribution, and

Australian pied stilt and poaka sampling sites in Australia and Aotearoa New Zealand (maps

not to scale).

2.2 Genotyping-by-sequencing

GBS was selected as an appropriate reduced-representation sequencing method based on cost

and availability within Aotearoa New Zealand (Galla et al. 2016). Of the available DNA

extracts, 145 samples (130 kakī, six pied stilts, and nine hybrids) had DNA of sufficient

quality and quantity as per GBS recommendations. This included 74 males and 62 females,

with nine individuals of unknown sex (Supplementary Table 1). Nineteen individuals

included in the previous genetic analysis by Steeves et al. (2010) had gDNA of sufficient

quantity and quality for inclusion here (Supplementary Table 1). We prepared two 96-well

plates of samples for GBS, containing 145 samples, two negative controls (DNA-free

controls) per plate, and three positive controls (replicate samples) across plates. We diluted

samples with gDNA concentration > 150 ng/μL to 100 ng/μL, and supplied ~1 μg of DNA

for all samples. GBS optimisation and sequencing was conducted by AgResearch Ltd.. The

optimised GBS protocol used a double-digest with restriction enzymes PstI-MspI. A single

library was generated for all 145 samples and controls with a fragment length filter of 193–

500 bp including adapter sequences. This library was sequenced on one lane of Illumina

HiSeq 2500 v4 sequencing for 101 cycles.

2.3 Reference-guided variant discovery

We assessed sequence quality with FastQC v0.11.5 (Andrews, 2010), and confirmed absence

of contamination in negative controls through BLAST searches against the nucleotide

database (Altschul et al., 1990). To provide accurate SNP discovery, we mapped GBS data to

the kakī reference genome (Galla et al., 2019). We demultiplexed and filtered raw sequences

with Sabre v1.0 (Joshi, 2013) and adapter trimmed with Cutadapt v1.17 (Martin, 2011). We

indexed genomes with BWA v0.7.17 (Li & Durbin, 2009), and mapped the processed GBS

reads to the genomes with BWA-MEM. We pre-processed the mapped reads for variant

discovery by adding read-group information with SAMtools, marking duplicates with Picard

v2.18.0 (Picard Toolkit, 2019), and realigning indels with the Genome Analysis Toolkit

(GATK) v3.5 (McKenna et al., 2010).

We compared five independent pipelines for reference-guided variant discovery. The

'GATK' pipeline used GATK's HaplotypeCaller and GenotypeGVCFs to call variants. The

'Samtools' pipeline used SAMtools v1.7 mpileup and BCFtools v1.6 variant caller (Li,

2011). The 'Platypus' pipeline used the callVariants tool in Platypus v0.8.1 (Rimmer et al.,

2014) with minimum mapping quality of 20, minimum base quality of 20, minimum depth to

call a variant of 2, and flag to generate indels set. The 'Stacks' pipeline implemented Stacks

v2.2 (Catchen et al., 2013) reference-guided pipeline with default parameters. The mapped

sequence reads were passed as input to GATK, Samtools, Platypus, and Stacks. The fifth

pipeline, 'Tassel', was run independently with the raw multiplexed GBS data passed as input

to TASSEL5-GBS2 v5.2.39 (Glaubitz et al., 2014). Tags were extracted from the data set

with a minimum quality score of 10, and then passed to BWA v0.7.12 for alignment against

the reference genome. The resulting SAM file was passed back to TASSEL5-GBS2 for

variant discovery with default settings, and all SNPs with quality ≥ 10 were retained.

2.4 Variant processing

By using a reference-guided approach, genomic location data was available for all variants,

and so we could compare variants produced across the five pipelines using VCFtools v0.1.15

(Danecek et al., 2011) *vcf-compare* following standardisation of variant call format files with

vcf-convert. We visualised the intersections of common variants among pipelines with the

package UpSetR (Conway et al., 2017; Lex et al., 2014) implemented in R v3.5.1 (R Core

Team, 2018). To improve confidence that the SNPs discovered were true SNPs rather than

the result of sequencing or mapping error, we produced a single variant set comprising all

variants detected via at least three pipelines from the intersections of variants common to

multiple pipelines generated using VCFtools *vcf-isec*, *vcf-merge* and *vcf-sort*. To produce a

set of biallelic SNPs to investigate admixture between kakī and poaka, we removed indels

and multiallelic SNPs from the composite variant set using VCFtools. To confirm absence of

contamination and replicability of lab processes, preliminary filtering tests and downstream

analyses had retained negative and positive controls, and once confirmed, these controls were

removed. We then excluded sites with $> 10\%$ missing data, a minor allele frequency < 0.01,

and a minimum quality score < 20. SNPs with mean depth over all individuals between $5\times$

and $200\times$ were retained, and individuals with $> 50\%$ missing data across all sites were

excluded (final n = 140). Due to the putative nature of the sex chromosomes in the kakī

reference genome, preliminary analyses were conducted both including and excluding SNPs

located on putative sex chromosomes. In addition, we subsampled the individuals to include

only a single individual from known family groups based on pedigree data, and excluded

known hybrid offspring. No differences in the extent of introgression was observed from

downstream analyses for any of these approaches (data not shown), but we conservatively

excluded SNPs located on putative sex chromosomes.

2.5 Discriminant Analysis of Principal Components

In an exploratory multivariate approach to population clustering, we conducted Discriminant

Analysis of Principal Components (DAPC) with adegenet v2.1.1 (Jombart, 2008; Jombart et

al., 2010; Jombart & Ahmed, 2011; Jombart & Collins, 2015b) in R v3.5.1. DAPC attempts

to partition variance in a between-group and within-group manner to maximise the

discrimination between groups. Using a multivariate approach allows for fine-scale

assessment of population structure, without relying on population genetic models, and so is

independent of the assumptions of HWE or linkage equilibrium associated with population

structuring analyses (Jombart et al., 2010). DAPC uses *a priori* information of the number of

clusters present in the data set (two, kakī and pied stilts including poaka), and then assesses

the discriminants that best explain those clusters. To prevent overfitting of the data, we

optimised DAPC parameters using the Bayesian Information Criterion (BIC) and *a*-scores,

and performed cross-validation following (Jombart & Collins, 2015a). The *a*-score measures

the trade-off between the power of discrimination and potential to overfitting the data, using a

randomisation of the data to determine when cluster assignment is successful due to the

analysis or due to random discrimination, and penalises the reassignment score by the number

of retained principal components (PCs). Cross-validation confirmed the appropriate numbers

of PCs, using a random seed to produce 1,000 replicate runs with a training set of 80% of the

data across up to sixty PCs. The accuracy of the retained PCs was then tested with the

remaining 20% of the data, and the PCs retained for the final DAPC were based on that

which produced the lowest mean squared error and highest mean success. The optimised

DAPC analysis was visualised to infer species differentiation and individual clustering. We

also implemented PCA and additional clustering analyses with the package SNPRelate

v1.20.1 (Zheng et al., 2012) in R v3.6.3 for comparison with DAPC and ADMIXTURE

analyses. Results produced with SNPRelate were concordant with those of DAPC and

ADMIXTURE (Supplementary File 1).

2.6 Analysis of introgression with ADMIXTURE

317
To estimate individual assignment to population clusters and to detect introgression, we

analysed each SNP set with a maximum likelihood method implemented in ADMIXTURE

v1.3.0 (Alexander et al., 2009). To minimise stochasticity across multiple runs, we conducted

100 iterations of ADMIXTURE analysis with each SNP set for $K = 1-6$, where K represents

the hypothesised number of population clusters, using a random seed, ten-fold CV, and with

point estimation terminating when the change in log-likelihood increased by < 0.0001. The

range of *K*-values was selected independently of the results of DAPC analysis, allowing for

differentiation between the two species (kakī and pied stilts), along with potential population

structuring among kakī, or differentiation between Australian pied stilts and poaka. To

determine the most appropriate value of *K* for each SNP set, we averaged CV error across the

100 iterations and visualised the results, with the lowest CV error representing the most likely

K.

There are two potential introgression scenarios that are of particular concern for kakī

conservation managers. In the first scenario, a substantial amount of introgression would be

identified among a small number of kakī due to recent undetected hybridisation in the wild.

Accurate identification of any such individuals will inform breeding management decisions

where the goal is to avoid genetic mixing between kakī and poaka. In addition, changes in

wild population management may be required to detect such rare hybridisation events. In the

second scenario, a moderate amount of introgression previously undetected with genetic

markers may be observed across a greater number of kakī, resulting from the presence of a

small number of dark hybrids among the founding individuals in the captive breeding

programme (Galla et al., 2020), which may have been exacerbated by subsequent selection

maintaining introgressed genes. In this second scenario, breeding management practices may

be altered to take into account the proportion of introgression in individuals to minimise

genetic admixture. In both scenarios, robust and accurate detection of introgression is

requiredto determine future research and management needs. Thus we defined a stringent

threshold of

> 0.95 individual probability to assign individuals as kakī, informed by the strong genetic

differentiation between these two species (Steeves et al., 2010).

We visualised mean assignment probabilities (Q-values) across all iterations with pophelper

v2.3.0 (Francis, 2017) in R v3.5.1. We used pophelper for file conversion for input to

CLUMPP to handle label switching. Consensus Q-values for each individual were calculated

with the Greedy algorithm over 100 iterations in CLUMPP vMacOSX 1.1.2 (Jakobsson &

Rosenberg, 2007), and we visualised the results with pophelper in R v3.5.1. We manually

assessed the final Q-matrix for all individuals using the predefined assignment threshold to

assign individuals as kakī. To reduce any potential influence of related individuals from the

ADMIXTURE analysis, the SNP set was filtered to exclude known related kakī individuals

based on pedigree data (a single individual was retained among each known family group).

This SNP set also excluded poaka to avoid any influence of potential admixture from kakī

among those individuals. ADMIXTURE and DAPC analyses were conducted as above, and

revealed no differences in population structure or patterns arising from family relatedness or

inclusion of poaka (Supplementary Figures S1 and S2).

2.7 Combining pedigree data with genomic population assignment data

Following all admixture and population clustering analyses, only six node J individuals were

identified with < 1.00 kakī assignment. A small number of hybrid individuals are included

among the founders in the kakī pedigree (Galla et al., 2020). To determine whether these

assignment probabilities could be attributed to known hybrid ancestry (< 7 generations deep),

we used the kakī pedigree (Galla et al., 2020) to assess the ancestry of these six individuals.

3 Results

Following a reference-guided multi-pipeline approach to SNP discovery and filtering for 145

stilts, a total of 140,948 SNPs were used in downstream analyses which detected no evidence

of introgression from poaka into kakī. Of the 250 gDNA samples available, 145 extractions

contained DNA of the required quantity and quality for GBS, including 66 of the 106

(63.2%) adults alive in the wild kakī population when this study began in 2017. GBS of the

pooled set of kakī, Australian pied stilts and poaka, and interspecific hybrids produced a total

of 303,639,199 raw sequences with length 35–101 bp and high sequence quality.

Demultiplexing produced an average of 2,024,530 ± SD 1,031,208.21 reads per sample

(Supplementary Table 2), and no samples failed to sequence. Negative controls produced a

low number of reads (mean = 2,585 ± SD 1554.76 reads per negative). Contamination checks

of negative controls produced no matches to the BLAST nucleotide database.

3.1 Variant discovery and filtering

Mapping of trimmed, filtered reads for all samples to the reference kakī genome produced an

average of 1,138,306.05 ± SD 597,406.95 mapped reads per individual (Table 2). This

represents an average of 85.4% reads per sample successfully mapped to the kakī reference

genome.

Table 2 Sequencing outputs and mapping success of GBS data from kakī, Australian pied

stilts/poaka, and interspecific hybrids averaged by species. Overall includes all samples along

with negative and positive controls.

	Mean reads	Mean cleaned reads	Mean reads mapped to kakī genome
Kakī (n = 130)	2,094,197.47	1,378,947.26	1,211,024.32

Pied stilts (n = 6)	1,722,470.33	966,686.33	879,153.17
Hybrids (n = 9)	1,181,117.00	812,732.00	732,418.11
Overall	1,971,321.13	1,293,796.12	1,138,306.05

Variant calling with GATK produced the fewest variants (35,441), while SAMtools produced

the most (488,940, Fig. 2). There were 177,437 variants common to ≥ 3 pipelines (Fig. 2).

Despite GATK producing the fewest variants among the five pipelines, the majority (92.68%)

were retained in the common variant set. SAMtools had the lowest proportion (35.14%) of

discovered variants retained in the common set. A total of 15,851 SNPs remained after

filtering. The five individuals that had produced the fewest raw sequences (26,725–313,884

reads) were subsequently excluded due to high (> 50%) levels of missing data. The total

genotyping rate was 96.91%.

Exploratory statistics indicated an even distribution of SNPs throughout the genome, with a

mean density of $0.014 \pm$ SD 0.158 SNPs/kb (Table 3). Mean depth of coverage per individual

was 11.663×. Kakī were well-differentiated from pied stilts with mean $F_{ST} = 0.622$ (Table 3).

Per-population summary statistics identified pied stilts as having higher diversity in terms of

nucleotide divergence, a greater number of variant sites, and more private alleles than either

kakī or hybrids (Table 4). Kakī displayed the lowest nucleotide diversity and fewest

polymorphic sites among the predefined groups (Table 4).

3.2 Discriminant Analysis of Principal Components

DAPC analysis identified Australian pied stilts and poaka as clustering separately from kakī,

with hybrids intermediate to the two species, though grouping more closely with kakī than

poaka (Fig. 3). Two node I/J hybrid individuals (DNA777 and DNA779) were found to

cluster with kakī. These results were concordant with clustering analyses conducted with

SNPRelate (Supplementary File 1)

Figure 2 UpSetR plot of the intersections of the total variants discovered from GBS data for kakī, Australian pied stilts and poaka, and interspecific

hybrids across five variant discovery pipelines: GATK, Platypus, Samtools, Stacks, and Tassel. Bottom left bars represent the total number of

variants discovered with each pipeline, while the main bar plot represents the number of variants common to multiple pipelines as indicated by the

filled circles below.

Table 3 Summary statistics for single-nucleotide polymorphisms (SNPs) produced from GBS data for stilts. Mean ± standard deviation (SD). KB = kilobase, HWE = Hardy-Weinberg Equilibrium, FDR = False Discovery Rate, F_{ST} = measure of population differentiation, Ts/Tv = ratio of transitions to transversions.

	Stilt SNP metrics
Total SNPs	15,851
Total samples	140
Total Kakī / Pied / Hybrid individuals	125 / 6 / 9
Mean depth per SNP per individual	11.663 ± 6.107
Mean per SNP depth	1,632.77 ± 668.176
Mean SNP quality	1,248.19 ± 2447.43
Mean frequency of missing data per individual	0.031 ± 0.073
Mean SNPs/KB	0.014 ± 0.158
Total singletons/private doubletons	0
Mean nucleotide diversity; π	0.0912 ± 0.100
SNPs deviating from HWE (FDR-corrected P ≤ 0.05)	758
Ts/Tv ratio	3.894
Weir & Cockerham mean F_{ST} (Kakī v Pied)	0.622
Weir & Cockerham weighted mean F_{ST} (Kakī v Pied)	0.637

Table 4 Population summary statistics from the three single-nucleotide polymorphism (SNP) sets produced from GBS data for kakī, Australian pied stilts/poaka, and interspecific hybrids, as calculated during format conversion from VCF to PLINK.

	Kakī	Pied	Hybrid
Mean samples per locus	121.990	5.542	8.141
Polymorphic sites	6,729	14,699	13,621
Private alleles	389	1,100	122

Mean nucleotide diversity (π)	0.057	0.396	0.235

Figure 3 Scatterplot of a Discriminant Analysis of Principal Components (DAPC) produced from GBS data for kakī, Australian pied stilts and poaka, and interspecific hybrids conducted in adegenet. Individual points are coloured according to plumage node and/or known ancestry as one of: kakī (*Himantopus novaezelandiae*), Australian pied stilts/poaka (*H. himantopus leucocephalus*), or interspecific hybrids. The closer individuals are to one another in the plot, the more likely they are to have shared genetic ancestry. DAPC analysis was optimised using *a*-score, cross-validation, and BIC to derive the appropriate number of principal components (2 discriminants, 10 PCs). The 67% inertial ellipses around each cluster represent the variance of the clusters depicted. The insert of PCA eigenvalues represents the variation explained by the PCs, and the insert of DA eigenvalues represents the magnitude of variation explained by the two discriminants.

3.3 ADMIXTURE analysis of introgression

$K = 2$ was indicated as the most likely number of clusters for ADMIXTURE analysis based

on CV error values (Supplementary Table 3), consistent with previous genetic results

confirming taxonomic delimitation (Steeves et al., 2010), and therefore only results for $K = 2$

for are reproduced here. All individuals categorised as kakī (node J individuals) had

assignment probabilities to the kakī cluster above the pre-defined 0.95 threshold, and only six

individuals were assigned as kakī with probability < 1.00 (kakī mean $Q = 0.9992 \pm$ SE

0.0009; Fig. 4, Table 5, Supplementary Table 4). Both Australian pied stilts and the poaka

individual Poaka1 were assigned with 1.00 probability to the pied stilt cluster. Assignment

probabilities to the kakī cluster for hybrids ranged from 0.2301 (DNA2113) to 1.00 (DNA777

and DNA779; hybrid mean $Q = 0.6399 \pm$ SD 0.0175; Supplementary Table 4).

459
Figure 4 Assignment probabilities for kakī (K), Australian pied stilts and poaka (P), and interspecific hybrids (H) produced via pophelper

visualisation of CLUMPP-permuted ADMIXTURE results when $K = 2$. Each individual is represented by a vertical bar, with colours indicating

the assignment probability to the kakī (green) or Australian pied stilts/poaka (grey) cluster.

Table 5 Mean individual assignment probabilities (Q-values) produced through 100 iterations

of ADMIXTURE analysis for kakī (node J individuals with known node J ancestry) that were

assigned to the kakī cluster with probabilities above the 0.95 threshold, but below 1.00.

DNA ID	Mean assignment probability, Q
DNA1252	0.9514
DNA1429	0.9838
DNA1620	0.9882
DNA1694	0.9945
DNA897	0.9882
DNA932	0.9910

3.4 Combining pedigree and genomic population assignment data

All node J individuals were assigned as kakī with probabilities above the 0.95 threshold.

Pedigree assessment revealed only 17 kakī individuals included in this study had no recorded

hybrid ancestry (i.e., all of their recorded ancestors were individuals with plumage node J

representing completely black birds). Among the node J kakī individuals identified as having

0.9500–0.9999 probability of assignment to the kakī cluster, all individuals had at least one

node I or I/J individual in their recorded ancestry. Individual DNA897 had the most recent

hybrid ancestry, with an I/J ancestor three generations deep. Individual DNA1252 had the

lowest probability of assignment to the kakī cluster among all kakī (Table 5), and while this

individual did not have more frequent or more recent recorded hybrid ancestry than other

kakī here, there is no recorded ancestry for the paternal lineage. The mother of individual

DNA1252 was included in sequencing and analyses, and was not identified as having any

hybrid ancestry, suggesting a paternal lineage origin for the relatively lower probability of

assignment as kakī. Individual DNA1694 has a relatively deep pedigree among kaki,

spanning seven generations. This individual also has the most frequent incidence of recorded

hybrid ancestry among these individuals, with a node I/J ancestor four generations deep, three

further I/J ancestors six generations deep, and a node I ancestor seven generations deep. The

two hybrid individuals that were consistently assigned to the kakī cluster were siblings

DNA777 and DNA779, with a node I/J mother, and a node I/J ancestor three generations

deep.

4 Discussion

With the development of genomic sequencing and associated methodologies, conservation

management of threatened species can now incorporate high-resolution genomic tools to

more robustly explore threats to species recovery, such as anthropogenic hybridisation

(Allendorf et al., 2010; Avise, 2010; Ekblom & Galindo, 2011; Gompert, 2012; Primmer,

2009). As discussed by Allendorf et al., (2001), hybridisation of a threatened species with a

more common non-native species can cause widespread introgression and the formation of a

hybrid swarm replacing the distinct species. Studies to date indicate that while using a small

number of genetic markers can provide important baseline data for conservation management

(e.g., genetic sexing, (Steeves et al., 2010); brood parasitism, (Overbeek et al., 2017;

Overbeek et al., 2020), such marker sets have proved less robust than genomic markers for

analyses of parentage (Tokarska et al., 2009), relatedness (Galla et al., 2020), intraspecific

population structure (McCartney-Melstad et al., 2018), and introgression (Parejo et al., 2018).

4.1 Impacts of hybridisation in stilts

All 130 kakī genotyped here—representing 63% of the contemporary adult population at the

time of sampling—were assigned as kakī with > 0.95 probability. These data confirm that

despite hybridisation with poaka, genetic admixture into kakī has not occurred, concordant

with earlier genetic analysis (Steeves et al., 2010). These results contrast with those of the

study by Toews et al. (2016) in which analysis of GBS data could not distinguish between

two hybridising species of warblers, and whole-genome sequencing data was required. This is

likely due to the very low differentiation observed between the two warbler species,

compared with the strong differentiation between kakī and pied stilts.

No individuals with pure-black plumage (node J) were identified as cryptic hybrids, with all

individuals assigned to the kakī cluster with probabilities above the predefined 0.95

threshold. Among the sampled individuals, 29 kakī had at least one hybrid ancestor three

generations in the past, and these ancestors were all dark hybrids (node I or I/J). No kakī

individuals included in this study had recorded hybrid ancestors with plumage lighter than

node I. These results indicate that with every generation of kakī backcrossing following

hybridisation, the proportion of introgressed genes is increasingly likely to be replaced by

kakī genetic material. Indeed, two individuals identified as hybrids (DNA777 and DNA779)

with a node I/J mother were assigned as kakī with probability of 1.00 across all analyses. This

may indicate that even a small number of generations of backcrossing with kakī can rapidly

overwhelm introgression from poaka. The current kakī recovery strategy excludes any non-

node J birds from active management (i.e., excluding any offspring of node I/J birds from the

captive rearing programme; Steeves et al., 2010).

The finding of no introgression from poaka to kakī likely results from a combination of

factors. First, the management strategy enacted by the DOC's Kakī Recovery Programme to

avoid genetic admixture of kakī has successfully minimised opportunities for hybridisation

between these species. This management has been responsive to the results of genetic

analysis that led to the exclusion of all non-node J individuals from conservation

management (Steeves et al., 2010). Intensive population monitoring of breeding pairs and

assessment of putative hybrids using the microsatellite panel has enabled practitioners to

break up mixed pairs (allowing kakī to re-pair with kakī) and exclude hybrids from the

captive breeding programme (Maloney & Murray, 2001). Ongoing kakī recovery has resulted

in a relatively balanced sex ratio in the wild, and combined with the strong positive

assortative mating of kakī, has minimised the likelihood of kakī breeding with non-kakī

(Steeves et al., 2010). Moderate outbreeding depression and stochastic processes have also

contributed to reduce the reproductive success of hybrids (Steeves et al., 2010), further

limiting the likelihood of introgressed material to be maintained in the population.

4.2 Implications for kakī conservation

The results presented here provide evidence that active conservation management designed to

minimise hybridisation can be effective in maintaining species integrity, and support the

ongoing management strategy of poaka and hybrid exclusion from the captive management

programme based on the results of genetic analysis (Steeves et al., 2010). The goals of

current kakī management to avoid genetic admixture are appropriate due to the strong

differentiation between species observed here, reduced fitness of hybrid offspring, and the

previous observation of moderate levels of genetic diversity among kakī compared with other

threatened Aotearoa New Zealand endemic birds (Steeves et al., 2010). Despite the increased

resolution of genomic data, when individuals with anomalous plumage are observed in the

captive breeding and rearing facility, microsatellite genotyping remains the most cost- and

time-efficient, low-complexity method for confirming species status (Table 6; Overbeek et

al., 2017; Overbeek et al., 2020), and this may be the case for other species threatened by

hybridisation. Early confirmation of hybrid status with genetic tools is key, as kakī do not

become fully black until about two years of age, although the Kakī Recovery Team are

experienced at identifying putative hybrid offspring appearing as unusually 'pale'

chicks/juveniles. Evidence of reduced fitness of hybrid offspring (Steeves et al., 2010)

suggests avoidance of genetic admixture will continue to be prioritised over any potential

gain in genetic diversity that may be facilitated through managed introgression (e.g., the

inclusion of cryptic hybrids). Further, the absence of cryptic hybrids in the contemporary kakī

population indicates continued avoidance of genetic admixture is possible for kakī (Steeves et

al., 2010; this study). One key challenge of hybridisation to conservation is consideration of

the impacts of inbreeding versus outbreeding (Liddell et al., 2021). Further research

pertaining to inbreeding and inbreeding depression could enhance understanding of the

consequences of maintaining these species as distinct entities, with particular focus on the

impacts of population decline and ongoing small population size on kakī genetic diversity as

compared with known impacts of outbreeding (Steeves et al., 2010). Nevertheless, the

mitigation of risks associated with inbreeding by the DOC's Kakī Recovery Programme, as

compared with the challenges of managing similar risks associated with outbreeding mean

the current management strategy remains most appropriate currently.

Under optimal circumstances, kakī recovery will continue, leading to increased numbers of

kakī within Te Manahuna in the short-term, and the potential for natural expansion beyond

the basin in the long-term. In the short-term recovery scenario, the success of the

conservation breeding and rearing programme to date may mean that active management of

the species could be scaled back. This may see a reduction in the number of adult kakī

maintained in captivity for breeding, although management of wild nests, including egg-

collection, artificial incubation and captive rearing are likely to continue to maximise

population growth while kakī remain critically endangered. As kakī are one of the few

threatened Aotearoa New Zealand birds to have maintained a population on the mainland

despite the presence of invasive predators, and are capable of travelling long distances, active

translocations are unlikely to be necessary to support natural expansion beyond Te

Manahuna. In addition, it is unlikely that active management of any such expansion would be

feasible. As such, management to minimise the likelihood of hybridisation within Te

Manahuna will continue, but with the wide distribution of poaka across the country, future

expansion into areas with high poaka densities may result in the increased prevalence of

hybridisation that may promote genetic admixture. Therefore, avoiding hybridisation in the

source population within Te Manahuna should be of high priority for conservation.

Table 6 Comparison of the costs and benefits associated with a genetic approach (i.e., microsatellite panel) for assessing hybridisation in kakī with

a genomics approach (i.e., genotyping-by-sequencing). With both platforms already established for kakī, cost per sample for additional small

number of samples assessed each year is the primary deciding factor. All cost estimates are in British pounds (GBP) based on cost estimates in

Aotearoa New Zealand.

	Microsatellite panel	**Genotyping-by-sequencing**
Platform development cost	< 5,000 GBP. Includes development and screening of ~20 polymorphic loci, and genotyping of up to 94 individuals (based on a known decrease in costs since the estimate of Galla et al. (2016)).	~3,000 GBP. Includes testing of restriction enzymes and adapter barcoding, plus DNA extraction and sequencing of up to 94 samples, with expected output of 30–60K SNPs discovered via a de novo approach to SNP discovery.
Platform development time	3–4 months	3–4 months
Cost per sample for additional samples	10 GBP. Includes DNA extraction and quantification, and microsatellite genotyping at eight optimised loci. Excludes associated person-hours.	30 GBP. Includes DNA extraction and quantification, and GBS. Excludes associated person-hours.
Lab time for additional samples	1 week	1–3 months
Analysis time	1 week (including manual genotyping)	4 weeks

Analysis requirements	Access to a standard desktop computer. Access to allele-calling software (e.g., GeneMarker v2.2). Access to and experience with genetic population clustering tools optional.	Access to a high-capacity computing system or capable computing cluster. Access to and experience with a variety of population genomic and bioinformatic tools.
Additional benefits	Previous uncertainty regarding how representative of genome-wide patterns of introgression these data are has been resolved for kakī by comparison with GBS data, indicating the eight loci are sufficient for robust identification of non-kakī individuals. All wet-lab work and analysis for additional samples can be run in-house.	Increased confidence in detection of introgression with large genome-wide SNP set compared with microsatellites. No ascertainment bias associated with marker generation. Once these data are generated, they can be implemented for a variety of downstream uses. Potential for these data to have additional application in the future with genomic analysis developments.
Limitations	Caveats associated with how the microsatellite loci are generated, and what they were developed for (e.g., active avoidance of regions under selection, ascertainment bias from selecting the most heterozygous markers). This limits the type of data produced, and thus the types of analyses that can be performed. Potential for human error associated with manual allele-calling, but mitigated with experience.	Additional sampling requires the SNP discovery pipeline to be run with all previous samples included every time, and so analysis does not become more efficient over time. Potential for batch effects between different sequencing batches (Leigh et al., 2018). Analyses limited by software and models available in a developing field.

4.3 Impacts of hybridisation on poaka

The identification of poaka with pied stilt assignment probabilities < 0.95 may be a result of initial small population size on arrival to Aotearoa New Zealand, and subsequent hybridisation with kakī prior to species decline. Kakī only occur as vagrants in the North Island of Aotearoa New Zealand, observed in very low numbers since at least the 1950s (Pierce, 1984b). Given the limited contact between kakī and poaka in the North Island in recent years, we expected the poaka samples sourced from the North Island to produce assignment probabilities similar to those of the Australian pied stilts. However, both individuals sourced from Auckland Zoo were assigned to the pied stilt cluster with < 0.95 probability. The only North Island individual with an assignment probability comparable to those of the Australian pied stilts was the individual from Hawke's Bay (Poaka1). This suggests that hybridisation early in the establishment of poaka may have resulted in introgression of kakī genetic material into an initially small poaka population that was not frequently supplemented by a substantial number of new immigrants, with introgressed material maintained in the expanding population despite subsequent backcrossing. Kakī introgression into poaka is supported by the observation of node A poaka having tarsal lengths outside the range observed among Australian pied stilts with no history of hybridisation, and poaka presenting a greater proportion of black plumage than is typical among Australian pied stilts (Pierce, 1984a).

4.4 Comparison of genetic and genomic approaches to introgression analysis

Reduced-representation sequencing approaches have proven to be efficient, robust, and cost-effective for variant discovery (Andrews et al., 2016; Davey et al., 2011; Elshire et al., 2011;

Peterson et al., 2012). Here GBS was used as a relatively cost-effective approach to

population-level genomic sequencing of non-model species, producing a set of species-

discriminating SNPs. Initial development of a GBS system is markedly less expensive than

development of a microsatellite panel (3000 GBP for GBS development using 94 samples in

this study in 2018 compared with 5000 GBP development and testing of a microsatellite

panel of approximately ten loci using 94 samples based on the estimate of Galla et al. (2016;

Table 6). However, ongoing costs of the microsatellite panel per sample remain considerably

lower than that of GBS (10 GBP/sample for a microsatellite panel compared with 30

GBP/sample for GBS; Table 6), and the time required from individual sampling to

completion of analysis is substantially reduced. There are also fewer barriers to analysing

microsatellite data (e.g., microsatellite genotyping and analyses can be conducted on a

standard desktop computing system compared with the requirement of a high-performance

computing system with access to and experience with a variety of bioinformatic tools

necessary to analyse GBS data, Table 6). While the ability to more readily characterise

genome-wide variation will make a genomics approach desirable for many conservation

projects, the associated costs may limit uptake, especially when providing data for time-

dependent decisions. Despite the increasing uptake of genomics approaches to answer

questions pertinent to conservation management (Galla et al., 2016), the current greater costs

and other transitional challenges (e.g., bioinformatic expertise) will likely maintain the

conservation genomics gap for at least some species for some time yet.

For kakī, the nature of GBS as a reduced-representation approach means that despite the large

increase in data compared with the eight microsatellite loci used previously, this only

represents < 1% of the 1.1 Gb kakī genome (Galla et al., 2019). While whole-genome

resequencing on the scale achieved here was prohibitively expensive when this study was

initiated, declining costs of resequencing have already overtaken reduced-representation

sequencing. For example, combined with the kakī pedigree, whole-genome resequencing data

is now being used to inform the kakī conservation breeding programme (Galla et al., 2020).

Thus, should renewed population decline and a skewed sex bias cause hybridisation to

increase and threaten kakī recovery in the future, or should the conservation value of cryptic

hybrids be reconsidered, then a comparison of GBS with whole-genome resequencing data

may be useful.

5 Conclusions

Studies comparing the utility of genetic and genomic approaches for generating estimates of

population genetic diversity and differentiation indicate that large SNP sets generally

outperform the small microsatellite sets typically used in conservation genetic studies

(Hauser et al., 2011; Hohenlohe et al., 2013; Santure et al., 2010; Weinman et al., 2015).

Thus, re-examining the extent of introgression between critically endangered kakī and non-

threatened congeneric poaka using a genomic approach was essential to ascertain the efficacy

of conservation management aimed at avoiding genetic admixture of kakī. While results are

concordant between genetic and genomic approaches for kakī and poaka, this may not be the

case for other species, particularly when hybridisation may be widespread (e.g., hybridisation

between koloa maoli/Hawaiian duck (*Anas wyvilliana*) and the invasive mallard (*A.*

platyrhynchos; Wells et al., 2019). Thus, we recommend that when genetic assessment has

not been conducted, or there is uncertainty as to whether genetic data have adequately

captured the impact of hybridisation, a genomic approach should be used. Further, we

suggest that when genetic and genomic results are concordant – which we anticipate will be

more likely for well-differentiated species – conservation managers can confidently continue

to use genetic tools, particularly when these remain more efficient and cost-effective.

6 Data accessibility

Kakī are a taonga (treasured) species for Māori (the Indigenous people of Aotearoa New

Zealand) and as such, genomic data derived from kakī are also recognised as taonga in their

own right. Due to the tapu (sacred) nature of these data, the data presented here are hosted on

a password-protected database at https://www.ucconsert.org/data and will be made available

at the discretion of the kaitiaki of the iwi (tribes) and hapū (subtribes) associated with kakī.

These data include raw genotyping-by-sequencing data, and the VCF comprising the

unfiltered SNP set. Code for the analyses described is available on GitHub at

https://github.com/natforsdick/Himantopus.

7 Acknowledgements

We extend our thanks to the tangata whenua who are kaitiaki for kakī, namely Te Rūnanga o

Ngāi Tahu, Te Ngāi Tūāhuriri Rūnanga, Te Rūnanga o Arowhenua, Te Rūnanga o Waihao,

and Te Rūnanga o Moeraki. This research was funded by a Royal Society of New Zealand

Rutherford Discovery Fellowship to MK, and NJF was supported by a University of Otago

Doctoral Scholarship. We are grateful to the Aotearoa New Zealand Department of

Conservation's Kakī Recovery Team, Richard Jakob-Hoff at Auckland Zoo, David

McLelland at Adelaide Zoo, and John Berry for assisting with provision of kakī, Australian

pied stilt, poaka, and interspecific hybrid samples for use in this project, along with

associated metadata. We thank our GBS provider AgResearch Ltd, along with thank Olga

703   **8 References**

705   Alexander, D. H., Novembre, J., & Lange, K. (2009). Fast model-based estimation of

706   ancestry in unrelated individuals. *Genome Research*, *19*(9), 1655–1664.

707   https://doi.org/10.1101/gr.094052.109

708   Allendorf, F. W., Hohenlohe, P. A., & Luikart, G. (2010). Genomics and the future of

709   conservation genetics. *Nature Reviews Genetics*, *11*(10), 697–709.

710   https://doi.org/10.1038/nrg2844

711   Allendorf, F. W., Leary, R. F., Spruell, P., & Wenburg, J. K. (2001). The problems with

712   hybrids: Setting conservation guidelines. *Trends in Ecology & Evolution*, *16*(11),

713   613–622. https://doi.org/10.1016/S0169-5347(01)02290-X

714   Altschul, S. F., Gish, W., Miller, W., Myers, E. W., & Lipman, D. J. (1990). Basic local

715   alignment search tool. *Journal of Molecular Biology*, *215*(3), 403–410.

716   https://doi.org/10.1016/S0022-2836(05)80360-2

717   Andrews, K. R., Good, J. M., Miller, M. R., Luikart, G., & Hohenlohe, P. A. (2016).

718   Harnessing the power of RADseq for ecological and evolutionary genomics. *Nature*

719   *Reviews Genetics*, *17*(2), 81–92. https://doi.org/10.1038/nrg.2015.28

720   Andrews, S. (2010). *FastQC: a quality control tool for high throughput sequence data.*

721   http://www.bioinformatics.babraham.ac.uk/projects/fastqc

722   Arnold, M. L. (1997). *Natural hybridization and species concepts*. Oxford University Press.

723   https://books.google.co.nz/books?hl=en&lr=&id=Jh8jVjEuDfUC&oi=fnd&pg=PR11

&ots=PbObh3Qp32&sig=fligYJ6xTvI37WapjEhBMlzlPVs&redir_esc=y#v=onepage

&q&f=false

Avise, J. C. (2010). Perspective: Conservation genetics enters the genomics era.

Conservation Genetics, *11*(2), 665–669. https://doi.org/10.1007/s10592-009-0006-y

Ba, H., Jia, B., Wang, G., Yang, Y., Kedem, G., & Li, C. (2017). Genome-wide SNP

discovery and analysis of genetic diversity in farmed sika deer (*Cervus nippon*) in

northeast China using double-digest restriction site-associated DNA sequencing. *G3:*

Genes, Genomes, Genetics, *7*(9), 3169–3176. https://doi.org/10.1534/g3.117.300082

BirdLife International. (2018). *Himantopus novaezelandiae* (amended version of 2016

assessment). *The IUCN Red List of Threatened Species*.

http://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T22693690A129560535.en

Booth, J. E., Gaston, K. J., Evans, K. L., & Armsworth, P. R. (2011). The value of species

rarity in biodiversity recreation: A birdwatching example. *Biological Conservation*,

144(11), 2728–2732. https://doi.org/10.1016/j.biocon.2011.02.018

Brumfield, R. T. (2010). Speciation genetics of biological invasions with hybridization.

Molecular Ecology, *19*(23), 5079–5083. https://doi.org/10.1111/j.1365-

294X.2010.04896.x

Catchen, J., Hohenlohe, P. A., Bassham, S., Amores, A., & Cresko, W. A. (2013). Stacks: An

analysis tool set for population genomics. *Molecular Ecology*, *22*(11), 3124–3140.

https://doi.org/10.1111/mec.12354

Chan, C. H., Ballantyne, K. N., Aikman, H., Fastier, D., Daugherty, C. H., & Chambers, G.

K. (2006). Genetic analysis of interspecific hybridisation in the world's only Forbes'

parakeet (*Cyanoramphus forbesi*) natural population. *Conservation Genetics*, *7*(4),

493–506. https://doi.org/10.1007/s10592-005-9060-2

Chen, N., Cosgrove, E. J., Bowman, R., Fitzpatrick, J. W., & Clark, A. G. (2016). Genomic

consequences of population decline in the endangered Florida scrub-jay. *Current*

Biology, *26*(21), 2974–2979. https://doi.org/10.1016/j.cub.2016.08.062

Chunco, A. J. (2014). Hybridization in a warmer world. *Ecology and Evolution*, *4*(10), 2019–

2031. https://doi.org/10.1002/ece3.1052

Combosch, D. J., Lemer, S., Ward, P. D., Landman, N. H., & Giribet, G. (2017). Genomic

signatures of evolution in Nautilus—An endangered living fossil. *Molecular Ecology*,

26(21), 5923–5938. https://doi.org/10.1111/mec.14344

Conway, J. R., Lex, A., & Gehlenborg, N. (2017). UpSetR: An R package for the

visualization of intersecting sets and their properties. *Bioinformatics*, *33*(18), 2938–

2940. https://doi.org/10.1093/bioinformatics/btx364

Courchamp, F., Angulo, E., Rivalan, P., Hall, R. J., Signoret, L., Bull, L., & Meinard, Y.

(2006). Rarity value and species extinction: The anthropogenic Allee effect. *PLOS*

Biology, *4*(12), e415. https://doi.org/10.1371/journal.pbio.0040415

Cunha, R. L., Forsman, Z. H., Belderok, R., Knapp, I. S. S., Castilho, R., & Toonen, R. J.

(2019). Rare coral under the genomic microscope: Timing and relationships among

Hawaiian Montipora. *BMC Evolutionary Biology*, *19*(1), 153.

https://doi.org/10.1186/s12862-019-1476-2

Danecek, P., Auton, A., Abecasis, G., Albers, C. A., Banks, E., DePristo, M. A., Handsaker,

R. E., Lunter, G., Marth, G. T., Sherry, S. T., McVean, G., & Durbin, R. (2011). The

variant call format and VCFtools. *Bioinformatics*, *27*(15), 2156–2158.

https://doi.org/10.1093/bioinformatics/btr330

Davey, J. W., & Blaxter, M. L. (2010). RADSeq: Next-generation population genetics.

Briefings in Functional Genomics, *9*(5–6), 416–423.

https://doi.org/10.1093/bfgp/elq031

Davey, J. W., Hohenlohe, P. A., Etter, P. D., Boone, J. Q., Catchen, J. M., & Blaxter, M. L.

(2011). Genome-wide genetic marker discovery and genotyping using next-generation

sequencing. *Nature Reviews Genetics*, *12*(7), 499–510.

https://doi.org/10.1038/nrg3012

Der Sarkissian, C., Ermini, L., Schubert, M., Yang, M. A., Librado, P., Fumagalli, M.,

Jónsson, H., Bar-Gal, G. K., Albrechtsen, A., Vieira, F. G., Petersen, B., Ginolhac, A.,

Seguin-Orlando, A., Magnussen, K., Fages, A., Gamba, C., Lorente-Galdos, B.,

Polani, S., Steiner, C., … Orlando, L. (2015). Evolutionary genomics and

conservation of the endangered Przewalski's horse. *Current Biology*, *25*(19), 2577–

2583. https://doi.org/10.1016/j.cub.2015.08.032

Draper, D., Laguna, E., & Marques, I. (2021). Demystifying negative connotations of

hybridization for less biased conservation policies. *Frontiers in Ecology and*

Evolution, 9. https://doi.org/10.3389/fevo.2021.637100

Dufresnes, C., & Dubey, S. (2020). Invasion genomics supports an old hybrid swarm of pool

frogs in Western Europe. *Biological Invasions*, *22*(2), 205–210.

https://doi.org/10.1007/s10530-019-02112-8

Edmands, S. (2007). Between a rock and a hard place: Evaluating the relative risks of

inbreeding and outbreeding for conservation and management. *Molecular Ecology*,

16(3), 463–475. https://doi.org/10.1111/j.1365-294X.2006.03148.x

Ekblom, R., & Galindo, J. (2011). Applications of next generation sequencing in molecular

ecology of non-model organisms. *Heredity*, *107*(1), 1–15.

https://doi.org/10.1038/hdy.2010.152

Elshire, R. J., Glaubitz, J. C., Sun, Q., Poland, J. A., Kawamoto, K., Buckler, E. S., &

Mitchell, S. E. (2011). A robust, simple genotyping-by-sequencing (GBS) approach

for high diversity species. *PLoS ONE*, *6*(5), e19379–e19379.

https://doi.org/10.1371/journal.pone.0019379

Estévez, R. A., Anderson, C. B., Pizarro, J. C., & Burgman, M. A. (2015). Clarifying values,

risk perceptions, and attitudes to resolve or avoid social conflicts in invasive species

management. *Conservation Biology*, *29*(1), 19–30. https://doi.org/10.1111/cobi.12359

Fischer, M. C., Rellstab, C., Leuzinger, M., Roumet, M., Gugerli, F., Shimizu, K. K.,

Holderegger, R., & Widmer, A. (2017). Estimating genomic diversity and population

differentiation – an empirical comparison of microsatellite and SNP variation in

Arabidopsis halleri. *BMC Genomics*, *18*(1), 69. https://doi.org/10.1186/s12864-016-

3459-7

Fitzpatrick, B. M., Johnson, J. R., Kump, D. K., Smith, J. J., Voss, S. R., & Shaffer, H. B.

(2010). Rapid spread of invasive genes into a threatened native species. *Proceedings*

of the National Academy of Sciences, *107*(8), 3606–3610.

https://doi.org/10.1073/pnas.0911802107

Francis, R. M. (2017). pophelper: An R package and web app to analyse and visualise

population structure. *Molecular Ecology Resources*, *17*(1), 27–32.

https://doi.org/10.1111/1755-0998.12509

Galla, S. J. (2019). Conservation genomic management of two critically endangered New

Zealand birds. PhD thesis. https://ir.canterbury.ac.nz/handle/10092/17759

Galla, S. J., Buckley, T. R., Elshire, R., Hale, M. L., Knapp, M., Mccallum, J., Moraga, R.,

Santure, A. W., Wilcox, P., & Steeves, T. E. (2016). Building strong relationships

between conservation genetics and primary industry leads to mutually beneficial

genomic advances. *Molecular Ecology*. https://doi.org/10.1111/mec.13837

Galla, S. J., Forsdick, N. J., Brown, L., Hoeppner, M., Knapp, M., Maloney, R. F., Moraga,

R., Santure, A. W., & Steeves, T. E. (2019). Reference genomes from distantly related

species can be used for discovery of single nucleotide polymorphisms to inform

conservation management. *Genes*, *10*(1), 9. https://doi.org/10.3390/genes10010009

Galla, S. J., Moraga, R., Brown, L., Cleland, S., Hoeppner, M. P., Maloney, R. F.,

Richardson, A., Slater, L., Santure, A. W., & Steeves, T. E. (2020). A comparison of

pedigree, genetic and genomic estimates of relatedness for informing pairing

decisions in two critically endangered birds: Implications for conservation breeding

programmes worldwide. *Evolutionary Applications*.

https://doi.org/10.1111/eva.12916

Glaubitz, J. C., Casstevens, T. M., Lu, F., Harriman, J., Elshire, R. J., Sun, Q., & Buckler, E.

S. (2014). TASSEL-GBS: a high capacity genotyping by sequencing analysis

pipeline. *PLoS ONE*, *9*(2), e90346–e90346.

https://doi.org/10.1371/journal.pone.0090346

Gompert, Z. (2012). Population genomics as a new tool for wildlife management. *Molecular*

Ecology, *21*(7), 1542–1544. https://doi.org/10.1111/j.1365-294X.2012.05471.x

Hagen, E. N., Hale, M. L., Maloney, R. F., & Steeves, T. E. (2011). Conservation genetic

management of a critically endangered New Zealand endemic bird: Minimizing

inbreeding in the Black Stilt *Himantopus novaezelandiae*. *Ibis*, *153*(3), 556–561.

https://doi.org/10.1111/j.1474-919X.2011.01137.x

Haig, S. M., & Allendorf, F. W. (2006). Hybrids and policy. In J. M. Scott, D. D. Goble, & F.

W. Davis (Eds.), *The Endangered Species Act at Thirty, Volume 2: Conserving*

Biodiversity in Human-Dominated Landscapes (pp. 150–163). Island Press.

https://digitalcommons.unl.edu/usgsstaffpub/708/

Hamilton, J. A., & Miller, J. M. (2016). Adaptive introgression as a resource for management

and genetic conservation in a changing climate. *Conservation Biology*, *30*(1), 33–41.

https://doi.org/10.1111/cobi.12574

Hanna, Z. R., Dumbacher, J. P., Bowie, R. C. K., Henderson, J. B., & Wall, J. D. (2018).

Whole-genome analysis of introgression between the Spotted Owl and Barred Owl

(*Strix occidentalis* and *Strix varia*, respectively; Aves: Strigidae) in Western North

America. *G3: Genes, Genomes, Genetics*, *8*(12), 3945–3952.

https://doi.org/10.1534/g3.118.200754

Hauser, L., Baird, M., Hilborn, R., Seeb, L. W., & Seeb, J. E. (2011). An empirical

comparison of SNPs and microsatellites for parentage and kinship assignment in a

wild sockeye salmon (*Oncorhynchus nerka*) population. *Molecular Ecology*

Resources, *11*(SUPPL. 1), 150–161. https://doi.org/10.1111/j.1755-

0998.2010.02961.x

Heezik, Y. van, Lei, P., Maloney, R. F., & Sancha, E. (2005). Captive breeding for

reintroduction: Influence of management practices and biological factors on survival

of captive kaki (black stilt). *Zoo Biology*, *24*(5), 459–474.

https://doi.org/10.1002/zoo.20065

Hirashiki, C., Kareiva, P., & Marvier, M. (n.d.). Concern over hybridization risks should not

preclude conservation interventions. *Conservation Science and Practice*, e424.

https://doi.org/10.1111/csp2.424

Hohenlohe, P. A., Day, M. D., Amish, S. J., Miller, M. R., Kamps-Hughes, N., Boyer, M. C.,

Muhlfeld, C. C., Allendorf, F. W., Johnson, E. A., & Luikart, G. (2013). Genomic

patterns of introgression in rainbow and westslope cutthroat trout illuminated by

overlapping paired-end RAD sequencing. *Molecular Ecology*, *22*(11), 3002–3013.

https://doi.org/10.1111/mec.12239

Jackiw, R. N., Mandil, G., & Hager, H. A. (2015). A framework to guide the conservation of

species hybrids based on ethical and ecological considerations. *Conservation Biology*,

29(4), 1040–1051. https://doi.org/10.1111/cobi.12526

Jakobsson, M., & Rosenberg, N. A. (2007). CLUMPP: A cluster matching and permutation

program for dealing with label switching and multimodality in analysis of population

structure. *Bioinformatics*, *23*(14), 1801–1806.

https://doi.org/10.1093/bioinformatics/btm233

Jombart, T. (2008). adegenet: A R package for the multivariate analysis of genetic markers.

Bioinformatics, *24*(11), 1403–1405. https://doi.org/10.1093/bioinformatics/btn129

Jombart, T., & Ahmed, I. (2011). adegenet 1.3-1: New tools for the analysis of genome-wide

SNP data. *Bioinformatics*, *27*(21), 3070–3071.

https://doi.org/10.1093/bioinformatics/btr521

Jombart, T., & Collins, C. (2015a). *A tutorial for Discriminant Analysis of Principal*

Components (DAPC) using adegenet 2.0.0. http://adegenet.r-forge.r-

project.org/files/tutorial-dapc.pdf

Jombart, T., & Collins, C. (2015b). *Analysing genome-wide SNP data using adegenet 2.0.0*.

http://adegenet.r-forge.r-project.org/files/tutorial-genomics.pdf

Jombart, T., Devillard, S., & Balloux, F. (2010). Discriminant analysis of principal

components: A new method for the analysis of genetically structured populations.

BMC Genetics, *11*(1), 94–94. https://doi.org/10.1186/1471-2156-11-94

Joshi, N. (2013). *Sabre—A barcode demultiplexing and trimming tool for FastQ files*.

https://github.com/najoshi/sabre

Kardos, M., Luikart, G., & Allendorf, F. W. (2015). Measuring individual inbreeding in the

age of genomics: Marker-based measures are better than pedigrees. *Heredity*, 115(1),

63–72. https://doi.org/10.1038/hdy.2015.17

Keedwell, R. J., Maloney, R. F., & Murray, D. P. (2002). Predator control for protecting kaki

(*Himantopus novaezelandiae*)—Lessons from 20 years of management. *Biological*

Conservation, *105*(3), 369–374. https://doi.org/10.1016/S0006-3207(01)00220-8

Lex, A., Gehlenborg, N., Strobelt, H., Vuillemot, R., & Pfister, H. (2014). UpSet:

Visualization of intersecting sets. *IEEE Transactions on Visualization and Computer*

Graphics, *20*(12), 1983–1992. https://doi.org/10.1109/TVCG.2014.2346248

Li, H. (2011). A statistical framework for SNP calling, mutation discovery, association

mapping and population genetical parameter estimation from sequencing data.

Bioinformatics, *27*(21), 2987–2993. https://doi.org/10.1093/bioinformatics/btr509

Li, H., & Durbin, R. (2009). Fast and accurate short read alignment with Burrows-Wheeler

transform. *Bioinformatics*, *25*(14), 1754–1760.

https://doi.org/10.1093/bioinformatics/btp324

Li, R., Fan, W., Tian, G., Zhu, H., He, L., Cai, J., Huang, Q., Cai, Q., Li, B., Bai, Y., Zhang,

Z., Zhang, Y., Wang, W., Li, J., Wei, F., Li, H., Jian, M., Li, J., Zhang, Z., … Wang,

J. (2010). The sequence and de novo assembly of the giant panda genome. *Nature*,

463(7279), 311–317. https://doi.org/10.1038/nature08696

Liddell, E., Sunnucks, P., & Cook, C. N. (2021). To mix or not to mix gene pools for

threatened species management? Few studies use genetic data to examine the risks of

both actions, but failing to do so leads disproportionately to recommendations for

separate management. *Biological Conservation*, 256, 109072.

https://doi.org/10.1016/j.biocon.2021.109072

Lynch, M. (1991). The genetic interpretation of inbreeding depression and outbreeding

depression. *Evolution; International Journal of Organic Evolution*, *45*(3), 622–629.

https://doi.org/10.1111/j.1558-5646.1991.tb04333.x

Ma, W., & Lambert, D. (1997). Minisatellite DNA markers reveal hybridisation between the

endangered black robin and tomtit. *ELECTROPHORESIS*, *18*(9), 1682–1687.

https://doi.org/10.1002/elps.1150180936

Maguire, L. A., & Justus, J. (2008). Why Intrinsic Value Is a Poor Basis for Conservation
Decisions. *BioScience*, *58*(10), 910–911. https://doi.org/10.1641/B581002

Mallet, J. (2008). Hybridization, ecological races and the nature of species: empirical
evidence for the ease of speciation. *Philosophical Transactions of the Royal Society
B: Biological Sciences*, 363(1506), 2971–2986.
https://doi.org/10.1098/rstb.2008.0081

Maloney, R. F., & Murray, D. P. (2001). *Kaki (black stilt) Recovery Plan 2001-2011*.
Department of Conservation, Wellington, New Zealand.

Martin, M. (2011). Cutadapt removes adapter sequences from high-throughput sequencing
reads. *EMBnet.Journal*, *17*, 10–12. http://dx.doi.org/10.14806/ej.17.1.200

Martin, C. H., Crawford, J. E., Turner, B. J., & Simons, L. H. (2016). Diabolical survival in
Death Valley: Recent pupfish colonization, gene flow and genetic assimilation in the
smallest species range on earth. *Proceedings of the Royal Society B: Biological
Sciences*, *283*(1823), 20152334. https://doi.org/10.1098/rspb.2015.2334

McCartney-Melstad, E., Vu, J. K., & Shaffer, H. B. (2018). Genomic data recover previously
undetectable fragmentation effects in an endangered amphibian. *Molecular Ecology*,
27(22), 4430–4443. https://doi.org/10.1111/mec.14892

McKenna, A., Hanna, M., Banks, E., Sivachenko, A., Cibulskis, K., Kernytsky, A.,
Garimella, K., Altshuler, D., Gabriel, S., Daly, M., & DePristo, M. A. (2010). The
Genome Analysis Toolkit: A MapReduce framework for analyzing next-generation
DNA sequencing data. *Genome Research*, *20*(9), 1297–1303.
https://doi.org/10.1101/gr.107524.110

Milián-García, Y., Ramos-Targarona, R., Pérez-Fleitas, E., Sosa-Rodríguez, G., Guerra-
Manchena, L., Alonso-Tabet, M., Espinosa-López, G., & Russello, M. A. (2015).
Genetic evidence of hybridization between the critically endangered Cuban crocodile

and the American crocodile: Implications for population history and in situ/ex situ

conservation. *Heredity*, *114*(3), 272–280. https://doi.org/10.1038/hdy.2014.96

Narum, S. R., Buerkle, C. A., Davey, J. W., Miller, M. R., & Hohenlohe, P. A. (2013).

Genotyping-by-sequencing in ecological and conservation genomics. *Molecular*

Ecology, *22*(11), 2841–2847. https://doi.org/10.1111/mec.12350

Overbeek, A. L., Hauber, M. E., Brown, E., Cleland, S., Maloney, R. F., & Steeves, T. E.

(2017). Evidence for brood parasitism in a critically endangered Charadriiform with

implications for conservation. *Journal of Ornithology*, *158*(1), 333–337.

https://doi.org/10.1007/s10336-016-1375-x

Overbeek, A. L., Galla, S. J., Cleland, S., Thyne, C., Maloney, R. F., & Steeves, T. E. (2020).

Pedigree validation using genetic markers in an intensively-managed taonga species,

the critically endangered kakī (*Himantopus novaezelandiae*). *Notornis*, *67*(4), 709–

716.Parejo, M., Henriques, D., Pinto, M. A., Soland-Reckeweg, G., & Neuditschko,

M. (2018). Empirical comparison of microsatellite and SNP markers to estimate

introgression in *Apis mellifera mellifera*. *Journal of Apicultural Research*, *57*(4), 504–

506. https://doi.org/10.1080/00218839.2018.1494894

Peek, R. A., Bedwell, M., O'Rourke, S. M., Goldberg, C., Wengert, G. M., & Miller, M. R.

(2019). Hybridization between two parapatric ranid frog species in the northern Sierra

Nevada, California, USA. *Molecular Ecology*, *28*(20), 4636–4647.

https://doi.org/10.1111/mec.15236

Peters, J. L., Lavretsky, P., DaCosta, J. M., Bielefeld, R. R., Feddersen, J. C., & Sorenson, M.

D. (2016). Population genomic data delineate conservation units in mottled ducks

(*Anas fulvigula*). *Biological Conservation*, *203*, 272–281.

https://doi.org/10.1016/j.biocon.2016.10.003

Peterson, B. K., Weber, J. N., Kay, E. H., Fisher, H. S., & Hoekstra, H. E. (2012). Double

digest RADseq: An inexpensive method for de novo SNP discovery and genotyping

in model and non-model species. *PLoS ONE*, *7*(5), e37135–e37135.

https://doi.org/10.1371/journal.pone.0037135

Picard Toolkit. (2019). *Picard Toolkit*. Broad Institute. http://broadinstitute.github.io/picard/

Pierce, R. J. (1984a). Plumage, morphology and hybridisation of New Zealand stilts

Himantopus spp. *Notornis*, *31*(2), 106–130.

Pierce, R. J. (1984b). The changed distribution of stilts in New Zealand. *Notornis*, *31*(1), 7–

18.

Pierpaoli, M., Birò, Z. S., Herrmann, M., Hupe, K., Fernandes, M., Ragni, B., Szemethy, L.,

& Randi, E. (2003). Genetic distinction of wildcat (*Felis silvestris*) populations in

Europe, and hybridization with domestic cats in Hungary. *Molecular Ecology*, *12*(10),

2585–2598. https://doi.org/10.1046/j.1365-294x.2003.01939.x

Primmer, C. R. (2009). From conservation genetics to conservation genomics. *Annals of the*

New York Academy of Sciences, *1162*, 357–368. https://doi.org/10.1111/j.1749-

6632.2009.04444.x

Quilodrán, C. S., Currat, M., & Montoya-Burgos, J. I. (2018). Effect of hybridization with

genome exclusion on extinction risk. *Conservation Biology*, *32*(5), 1139–1149.

https://doi.org/10.1111/cobi.13120

R Core Team. (2018). *R: A language and environment for statistical computing*.

https://www.R-project.org/

Reed, C. E. M., Butler, D., & Murray, D. P. (1993). *Black Stilt Recovery Plan (Himantopus*

novaezealandiae). Department of Conservation, Wellington, New Zealand.

Rexer-Huber, K., Veale, A. J., Catry, P., Cherel, Y., Dutoit, L., Foster, Y., McEwan, J. C.,

Parker, G. C., Phillips, R. A., Ryan, P. G., Stanworth, A. J., Stijn, T. van, Thompson,

D. R., Waters, J., & Robertson, B. C. (2019). Genomics detects population structure

within and between ocean basins in a circumpolar seabird: The white-chinned petrel.

Molecular Ecology, *28*(20), 4552–4572. https://doi.org/10.1111/mec.15248

Rhymer, J. M., & Simberloff, D. (1996). Extinction by hybridization and introgression.

Annual Review of Ecology and Systematics, *27*(1), 83–109.

https://doi.org/10.1146/annurev.ecolsys.27.1.83

Richardson, L., & Loomis, J. (2009). The total economic value of threatened, endangered and

rare species: An updated meta-analysis. *Ecological Economics*, *68*(5), 1535–1548.

https://doi.org/10.1016/j.ecolecon.2008.10.016

Rick, K., Ottewell, K., Lohr, C., Thavornkanlapachai, R., Byrne, M., & Kennington, W. J.

(2019). Population genomics of Bettongia lesueur: Admixing increases genetic

diversity with no evidence of outbreeding depression. *Genes*, *10*(11), 851.

https://doi.org/10.3390/genes10110851

Riley, S. P. D., Bradley Shaffer, H., Randal Voss, S., & Fitzpatrick, B. M. (2003).

Hybridization between a rare, native tiger salamander (*Ambystoma californiense*) and

its introduced congener. *Ecological Applications*, *13*(5), 1263–1275.

https://doi.org/10.1890/02-5023

Rimmer, A., Phan, H., Mathieson, I., Iqbal, Z., Twigg, S. R. F., Wilkie, A. O. M., McVean,

G., & Lunter, G. (2014). Integrating mapping-, assembly- and haplotype-based

approaches for calling variants in clinical sequencing applications. *Nature Genetics*,

46(8), 912–918. https://doi.org/10.1038/ng.3036

Robertson, H. A., Baird, K., Dowding, J. E., Elliott, G. P., Hitchmough, R. A., Miskelly, C.

M., McArthur, N., O'Donnell, C. F. J., Sagar, P. M., Scofield, R. P., & Taylor, G. A.

(2016). *Conservation status of New Zealand birds, 2016* (p. 27). Department of

Conservation, Wellington, New Zealand.

Santure, A. W., Stapley, J., Ball, A. D., Birkhead, T. R., Burke, T., & Slate, J. (2010). On the

use of large marker panels to estimate inbreeding and relatedness: Empirical and

simulation studies of a pedigreed zebra finch population typed at 771 SNPs.

Molecular Ecology, *19*(7), 1439–1451. https://doi.org/10.1111/j.1365-

294X.2010.04554.x

Schlaepfer, M. A., Sax, D. F., & Olden, J. D. (2011). The potential conservation value of

non-native species. *Conservation Biology*, *25*(3), 428–437.

https://doi.org/10.1111/j.1523-1739.2010.01646.x

Seabury, C. M., Bhattarai, E. K., Taylor, J. F., Viswanathan, G. G., Cooper, S. M., Davis, D.

S., Dowd, S. E., Lockwood, M. L., & Seabury, P. M. (2011). Genome-wide

polymorphism and comparative analyses in the white-tailed deer (*Odocoileus*

virginianus): A model for conservation genomics. *PloS One*, *6*(1), e15811.

https://doi.org/10.1371/journal.pone.0015811

Sovic, M. G., Fries, A. C., & Gibbs, H. L. (2016). Origin of a cryptic lineage in a threatened

reptile through isolation and historical hybridization. *Heredity*, *117*(5), 358–366.

https://doi.org/10.1038/hdy.2016.56

Steeves, T. E., Maloney, R. F., Hale, M. L., Tylianakis, J. M., & Gemmell, N. J. (2010).

Genetic analyses reveal hybridization but no hybrid swarm in one of the world's

rarest birds. *Molecular Ecology*, *19*, 5090–5100. https://doi.org/10.1111/j.1365-

294X.2010.04895.x

Sutton, J. T., Helmkampf, M., Steiner, C. C., Bellinger, M. R., Korlach, J., Hall, R.,

Baybayan, P., Muehling, J., Gu, J., Kingan, S., Masuda, B. M., & Ryder, O. A.

(2018). A high-quality, long-read de novo genome assembly to aid conservation of

Hawaii's last remaining crow species. *Genes*, *9*(8), 393.

https://doi.org/10.3390/genes9080393

Taylor, E. B., Boughman, J. W., Groenenboom, M., Sniatynski, M., Schluter, D., & Gow, J.

L. (2006). Speciation in reverse: Morphological and genetic evidence of the collapse

of a three-spined stickleback (*Gasterosteus aculeatus*) species pair. *Molecular*

Ecology, *15*(2), 343–355. https://doi.org/10.1111/j.1365-294X.2005.02794.x

Taylor, H. R. (2015). The use and abuse of genetic marker-based estimates of relatedness and

inbreeding. *Ecology and Evolution*, *5*(15), 3140–3150.

https://doi.org/10.1002/ece3.1541

Taylor, H. R., Kardos, M. D., Ramstad, K. M., & Allendorf, F. W. (2015). Valid estimates of

individual inbreeding coefficients from marker-based pedigrees are not feasible in

wild populations with low allelic diversity. *Conservation Genetics*, *16*(4), 901–913.

https://doi.org/10.1007/s10592-015-0709-1

Todesco, M., Pascual, M. A., Owens, G. L., Ostevik, K. L., Moyers, B. T., Hübner, S.,

Heredia, S. M., Hahn, M. A., Caseys, C., Bock, D. G., & Rieseberg, L. H. (2016).

Hybridization and extinction. *Evolutionary Applications*, *9*(7), 892–908.

https://doi.org/10.1111/eva.12367

Toews, D. P. L., Taylor, S. A., Vallender, R., Brelsford, A., Butcher, B. G., Messer, P. W., &

Lovette, I. J. (2016). Plumage genes and little else distinguish the genomes of

hybridizing warblers. *Current Biology*, *26*(17), 2313–2318.

https://doi.org/10.1016/J.CUB.2016.06.034

Tokarska, M., Marshall, T., Kowalczyk, R., Wójcik, J. M., Pertoldi, C., Kristensen, T. N.,

Loeschcke, V., Gregersen, V. R., & Bendixen, C. (2009). Effectiveness of

microsatellite and SNP markers for parentage and identity analysis in species with

low genetic diversity: The case of European bison. *Heredity*, *103*(4), 326–332.

https://doi.org/10.1038/hdy.2009.73

Väli, Ü., Einarsson, A., Waits, L., & Ellegren, H. (2008). To what extent do microsatellite

markers reflect genome-wide genetic diversity in natural populations? *Molecular*

Ecology, *17*(17), 3808–3817. https://doi.org/10.1111/j.1365-294X.2008.03876.x

vonHoldt, B. M., Brzeski, K. E., Wilcove, D. S., & Rutledge, L. Y. (2018). Redefining the

role of admixture and genomics in species conservation. *Conservation Letters*, *11*(2),

e12371. https://doi.org/10.1111/conl.12371

Wayne, R. K., & Shaffer, H. B. (2016). Hybridization and endangered species protection in

the molecular era. *Molecular Ecology*, *25*(11), 2680–2689.

https://doi.org/10.1111/mec.13642

Weinman, L. R., Solomon, J. W., & Rubenstein, D. R. (2015). A comparison of single

nucleotide polymorphism and microsatellite markers for analysis of parentage and

kinship in a cooperatively breeding bird. *Molecular Ecology Resources*, *15*(3), 502–

511. https://doi.org/10.1111/1755-0998.12330

Wells, C. P., Lavretsky, P., Sorenson, M. D., Peters, J. L., DaCosta, J. M., Turnbull, S.,

Uyehara, K. J., Malachowski, C. P., Dugger, B. D., Eadie, J. M., & Engilis, A. (2019).

Persistence of an endangered native duck, feral mallards, and multiple hybrid swarms

across the main Hawaiian Islands. *Molecular Ecology*, *28*(24), 5203–5216.

https://doi.org/10.1111/mec.15286

Wolf, D. E., Takebayashi, N., & Rieseberg, L. H. (2001). Predicting the risk of extinction

through hybridization. *Conservation Biology*, *15*(4), 1039–1053.

https://doi.org/10.1046/j.1523-1739.2001.0150041039.x

Wright, B. R., Grueber, C. E., Lott, M. J., Belov, K., Johnson, R. N., & Hogg, C. J. (2019).

Impact of reduced-representation sequencing protocols on detecting population

structure in a threatened marsupial. *Molecular Biology Reports*, 1–6.

https://doi.org/10.1007/s11033-019-04966

Zheng, X., Levine, D., Shen, J., Gogarten, S. M., Laurie, C., & Weir, B. S. (2012). A high-

performance computing toolset for relatedness and principal component analysis of

SNP data. *Bioinformatics*, 28(24), 3326–3328.

https://doi.org/10.1093/bioinformatics/bts606

Kakī X **Poaka**

Are there cryptic hybrids among kakī? **?**

We found no evidence of cryptic hybrids among kakī.