

Genomic analyses indicate resilience of a commercially and culturally important marine gastropod snail to climate change

Matt J. Nimbs^{Corresp., 1, 2}, Curtis Champion^{1, 2}, Simon E. Lobos^{3, 4}, Hamish A Malcolm⁵, Adam D. Miller^{3, 4}, Kate Seignor¹, Stephen D. A. Smith^{1, 6}, Nathan Knott⁷, David Wheeler⁸, Melinda A Coleman^{1, 2}

¹ National Marine Science Centre, 2 Bay Drive, Southern Cross University, Coffs Harbour, New South Wales, Australia

² National Marine Science Centre, 2 Bay Drive, New South Wales Fisheries, Coffs Harbour, NSW, Australia

³ School of Life and Environmental Sciences, Deakin University, Warrnambool, Vic, Australia

⁴ Deakin Genomics Centre, Deakin University, Geelong, Vic, Australia

⁵ NSW Department of Primary Industries, Fisheries Research, Coffs Harbour, NSW, Australia

⁶ Aquamarine Australia, Mullaway, NSW, Australia

⁷ NSW Department of Primary Industries, Fisheries Research, Huskisson, NSW, Australia

⁸ NSW Department of Primary Industries, Orange, NSW, Australia

Corresponding Author: Matt J. Nimbs

Email address: matthew.nimbs@dpi.nsw.gov.au

Genomic vulnerability analyses are being increasingly used to assess the adaptability of species to climate change and provide an opportunity for proactive management of harvested marine species in changing oceans. Southeastern Australia is a climate change hotspot where many marine species are shifting poleward. The turban snail, *Turbo militaris* is a commercially and culturally harvested marine gastropod snail from eastern Australia. The species has exhibited a climate-driven poleward range shift over the last two decades presenting an ongoing challenge for sustainable fisheries management. We investigate the impact of future climate change on *T. militaris* using genotype-by-sequencing to project patterns of gene flow and local adaptation across its range under climate change scenarios. A single admixed, and potentially panmictic, demographic unit was revealed with no evidence of genetic subdivision across the species range. Significant genotype associations with heterogeneous habitat features were observed, including associations with sea surface temperature, ocean currents, and nutrients, indicating possible adaptive genetic differentiation. These findings suggest that standing genetic variation may be available for selection to counter future environmental change, assisted by widespread gene flow, high fecundity and short generation time in this species. We discuss the findings of this study in the content of future fisheries management and conservation.

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6 ^{4,5}, Kate Seinor ², Stephen D. A. Smith ^{2,6}, Nathan A. Knott ⁷, David Wheeler ⁸, Melinda A.
7 Coleman ^{1,2}

8

9 ¹ Fisheries Research, NSW Department of Primary Industries, National Marine Science Centre, 2
10 Bay Drive, Coffs Harbour, NSW, 2450, Australia

11 ² National Marine Science Centre, Southern Cross University, 2 Bay Drive, Coffs Harbour,
12 NSW, Australia

13 ³ Fisheries Research, NSW Department of Primary Industries, Coffs Harbour, NSW, Australia

14 ⁴ School of Life and Environmental Sciences, Deakin University, Warrnambool Victoria,
15 Australia

16 ⁵ Deakin Genomics Centre, Deakin University, Geelong Victoria, Australia

17 ⁶ Aquamarine Australia, Mullaway, NSW, Australia

18 ⁷ Fisheries Research, NSW Department of Primary Industries, Huskisson, NSW, Australia

19 ⁸ NSW Department of Primary Industries, Orange, NSW, Australia

20

21 Corresponding Author:

22 Matt J. Nimbs¹

23 New South Wales Fisheries, National Marine Science Centre, 2 Bay Drive, Coffs Harbour,
24 NSW, 2450, Australia

25 Email address: matthew.nimbs@dpi.nsw.gov.au

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27 Abstract

28

29 Genomic vulnerability analyses are being increasingly used to assess the adaptability of species
30 to climate change and provide an opportunity for proactive management of harvested marine
31 species in changing oceans. Southeastern Australia is a climate change hotspot where many
32 marine species are shifting poleward. The turban snail, *Turbo militaris* is a commercially and
33 culturally harvested marine gastropod snail from eastern Australia. The species has exhibited a
34 climate-driven poleward range shift over the last two decades presenting an ongoing challenge
35 for sustainable fisheries management. We investigate the impact of future climate change on *T.*
36 *militaris* using genotype-by-sequencing to project patterns of gene flow and local adaptation

37 across its range under climate change scenarios. A single admixed, and potentially panmictic,
38 demographic unit was revealed with no evidence of genetic subdivision across the species range.
39 Significant genotype associations with heterogeneous habitat features were observed, including
40 associations with sea surface temperature, ocean currents, and nutrients, indicating possible
41 adaptive genetic differentiation. These findings suggest that standing genetic variation may be
42 available for selection to counter future environmental change, assisted by widespread gene flow,
43 high fecundity and short generation time in this species. We discuss the findings of this study in
44 the context of future fisheries management and conservation.

45

46 **Introduction**

47 Rapid climate change is impacting the physical state of the world's oceans and directly
48 threatening the structure and function of marine ecosystems. In particular, ocean warming and
49 marine heatwave events, ocean acidification and deoxygenation, changes to ocean currents, and
50 sea-level rise are already impacting many marine ecosystems and associated socioeconomic and
51 cultural values at a global scale (Brierley and Kingsford 2009; Doney et al. 2011; Orr et al. 2005;
52 Poloczanska et al. 2013, 2016; Scavia et al. 2002). Shifts in the physical ocean climate pose a
53 direct threat to many of the world's commercial fisheries, causing changes in species
54 distributions and abundances, altering habitats, decoupling critical trophic interactions, and
55 pushing species beyond their physiological limits (Holland et al. 2021; Hollowed et al. 2013;
56 Roessig et al. 2004; Sumaila et al. 2011). These effects are expected to reduce harvestable
57 biomass in many fisheries (Brander 2013), with studies predicting reductions in catch of up to
58 40% for tropical fisheries alone under the RCP8.5 scenario (Lam et al. 2020). Projections
59 suggest that climate change will continue to be an ongoing challenge for sustainable fisheries
60 management into the future (Cheung et al. 2010), highlighting the importance of research aimed
61 at understanding the resilience of individual fisheries to climate change effects, and identifying
62 interventions capable of 'climate-proofing' vulnerable fisheries (Fankhauser and Schmidt-Traub
63 2011; Harte et al. 2019).

64 Evidence suggests the ability of many marine species to track their thermal niche via migration is
65 likely to be outpaced by rapid climate change (Hiddink et al. 2012; Vranken et al. 2021). This
66 typically applies to less vagile organisms whose persistence will depend more on their ability to

67 adapt to new thermal environments, either through plasticity or genetic evolution (Donelson et
68 al. 2019; Munday et al. 2013; Somero 2010). Species with wide latitudinal ranges often show
69 genetically based clines across thermal gradients (Berger et al. 2013; Jenkins et al. 2019; Pereira,
70 Sasaki, and Burton 2017), suggesting standing genetic variation in quantitative traits might be
71 available for adaptation to new environments (Barrett and Schluter 2008; Sasaki et al. 2022). Yet
72 it is anticipated that selection itself may fail to keep pace with rapid climate change, particularly
73 in long-lived organisms (Vranken et al. 2021; Wood et al. 2021). In such cases, gene flow is
74 likely to play a critical role in the adaptation process, particularly when strong biological
75 connections among locally adapted populations from thermal environments are present (Miller et
76 al. 2019; Sork et al. 2010). Consequently, species with high connectivity among locally adapted
77 populations may be deemed less at risk than those species with limited gene flow and poor
78 dispersal capabilities (Ayre and Hughes 2004; Coleman 2015). In such cases, strategic
79 intervention measures may be needed to maximise the adaptive capacity of threatened fish
80 stocks, such as augmentation activities that include deliberately introducing genotypes from
81 warm adapted populations to those at risks of maladaptation (Aitken and Whitlock 2013;
82 Hagedorn et al. 2018; Layton et al. 2020; Hoffmann, Miller, and Weeks 2021). Consequently,
83 the challenge for fisheries managers is understanding when interventions of this nature will be
84 necessary in order to mitigate the risks of climate change.

85 Population genetics has played a key role in characterising structure and patterns of gene flow
86 among populations, especially in commercially important marine species over the last few
87 decades (Miller et al. 2019; Smith, Francis, and McVeagh 1991; Ward 2000; van Oppen and
88 Coleman 2022). This field of research has been revolutionised by modern DNA sequencing
89 technologies that now allows for genome wide estimates of genetic variation, providing
90 unprecedented power for resolving fine scale patterns of genetic structure (Cheng et al. 2021;
91 Milano et al. 2014; Sağlam et al. 2021) and signatures of adaptive genetic differentiation among
92 fishing stocks spanning environmental gradients (Quigley, Bay, and van Oppen 2020; Riquet et
93 al. 2013). Combined, this information can greatly assist fisheries management by providing
94 insights into the availability of standing variation for adaptation to future environmental
95 challenges, and the potential role of gene flow in assisting the adaptation process (Mason et al.
96 2022; Papa et al. 2020; Valenzuela-Quiñonez 2016). Importantly, this information can help to
97 identify fish stocks most at risk of maladaptation, and to guide stock augmentation programs

98 aimed at introducing novel genotypes for selection to act upon to help combat future
99 environmental challenges (Bernatchez et al. 2017; Reiss et al. 2009; Waples and Naish 2009).

100 Marine gastropod snails support many commercial and recreational fisheries around the world
101 (Dolorosa et al. 2010; Foale and Day 1997; Leiva and Castilla 2002), many of which are
102 expected to be susceptible to climate change effects (Ortíz, Arcos-Ortega, and Navarro 2022;
103 Valles-Regino et al. 2022). Specifically, evidence suggests that ocean warming has the potential
104 to suppress larval development, delay gonad maturation, reduce fecundity, stunt growth, and
105 increase susceptibility to disease in many species, while increased ocean acidification is expected
106 to compromise shell production in others (Holland et al. 2021; Leung, Russell, and Connell
107 2020; Zacherl, Gaines, and Lonhart 2003). However, some marine gastropods have broad
108 latitudinal distributions and show genetically determined clines across thermal gradients, and
109 strong biological connections over vast geographic distances due to long pelagic larval dispersal
110 stages (Kelly and Palumbi 2010; Miller et al. 2016; Neethling et al. 2008; Villamor, Costantini,
111 and Abbiati 2014). These findings suggest that some species might possess heritable genetic
112 variation for adaptation to future thermal environments, which maybe assisted by extensive gene
113 flow across thermal gradients (Johansson 2015; Sanford and Kelly 2010; Zardi et al. 2011).
114 However, information for some marine gastropod groups is still lacking and is urgently needed
115 to inform future management.

116 Benthic marine gastropods represent approximately 2% of the global marine mollusc fishery
117 (Department 1997, 2012), with some species having high economic value being targeted in
118 small-scale artisanal fisheries (i.e. *Haliotis* spp., *Strombus* spp., *Busycon* spp, and *Concholepas*
119 spp.). In recent decades commercial landings have grown with wild-stock catch increasing from
120 75k metric tonnes (mt) in 1979 to 103k mt in 1996 (Department 1997), however, some gastropod
121 fisheries have been identified as being significantly threatened by climate change (Carranza and
122 Arim 2023; Ramos et al. 2022). Unfortunately, information of the genetic structure and resilience
123 of many commercially important gastropods is lacking, including trochid snails. To date, only a
124 handful of studies have investigated patterns of population genetic structure among trochoid
125 fisheries (Berry et al. 2019; Díaz-Ferguson et al. 2010; Nikula, Spencer, and Waters 2011), and,
126 to our knowledge, no study has investigated patterns of adaptive genetic variation across the
127 entire range of any trochid species using a dataset that includes thousands of genome-wide
128 variants. *Turbo militaris* Reeve, 1848, is a large intertidal and shallow subtidal turbinid (Family

129 Turbinidae) from south-eastern Australian coast that is traditionally and recreationally harvested
130 for human consumption (Yearsley, Last, and Ward 1999) and supports a 6.6 tonne turban snail
131 commercial fishery in New South Wales (NSW) (DPI NSW 2019). Like many trochids, *T.*
132 *militaris* is showing signs of climate stress, with evidence of a poleward range shift since the turn
133 of the century (ALA 2018; GBIF 2016, Benkendorff & Przeslawski 2008). Given that range
134 shifts in commercially important marine species can have socio-economic and management
135 implications (Madin et al. 2012; Bonebrake et al. 2018), it is prudent to explore the capability of
136 *T. militaris* to adapt to ongoing environmental change in order to maintain extant populations.

137 In this study we investigated patterns of population genetic structure in *T. militaris* using
138 genotype-by-sequencing and a sampling regime encompassing the species' entire distribution
139 spanning seven degrees of latitude and a difference of 3.9°C in mean annual sea surface
140 temperature. We explore patterns of gene flow and population connectivity across the sampling
141 distribution and integrate genomic and geospatial data to tests for genotype-environmental
142 associations (GEAs) indicative of adaptive genetic differentiation between populations. Findings
143 from this study provide valuable insights into the spatial scales of gene flow and the availability
144 of standing genetic variation for adaptation to future environmental conditions. We discuss the
145 findings of this study in the context of future fisheries management.

146 **Materials & Methods**

147 *Sampling*

148 Eight rocky shore locations (0 to 5m depth) were selected for sampling (Table 1) spanning the
149 known range of *T. militaris* from Hastings Point (northern NSW) to Jervis Bay (southern NSW)
150 (Fig. 1), representing a seven-degree latitudinal and a 3.9°C annual mean sea surface temperature
151 gradient. Additionally, to sample a known cross-shelf gradient of temperature (less confounded
152 by distance and unconfounded by latitude), three locations at varying distances from shore and
153 under different temperature regimes were selected: Nambucca Heads (mainland), Split Solitary
154 Island (3 km from the mainland) and South Solitary Island (7 km from the mainland). These
155 represent inshore, mid-shelf, and offshore positions, respectively, with an annual mean sea
156 temperature varying by about 1°C from inshore to offshore (Malcolm, Jordan, and Smith 2010).
157 Notably, South and Split Solitary islands are remote from shore and are also likely to have very
158 low to no harvesting pressure. Between 25 and 30 *T. militaris* individuals were collected from

159 each location (Table 1). As internal foot muscle tissue was found to yield clean, high molecular-
160 weight DNA, it was necessary to narcotise specimens. Field collections were carried out under a
161 scientific collection permit granted by the NSW DPI Fisheries under section 37 of the NSW
162 Fisheries Management Act 1994 (#P01/0059(A)-4.0).

163 *DNA extraction and genotyping*

164 Approximately 25 mg of foot muscle tissue was sampled from each specimen using sterilised
165 scalpel and forceps, avoiding the inclusion of mucous-rich epidermal tissue. Tissue samples were
166 immediately placed in 2.0mL Eppendorf snap-lock microcentrifuge tubes containing 500µL of
167 hexadecyltrimethylammonium bromide (CTAB) lysis buffer (100mM TrisHCl, 20mM EDTA,
168 CTAB 2% w/v, NaCl 1.5M) and refrigerated at 4°C for two weeks. Proteinase K (30 µL at
169 20mg/mL) was added to the samples which were incubated in a Allsheng shaking incubator at
170 60°C, 200 rpm overnight. Samples were cooled to room temperature and purified by addition of
171 an equal volume of 24:1 chloroform isoamyl. After centrifugation, the aqueous phase was
172 retained and DNA was precipitated by the addition of 800 µL of dilute CTAB buffer (100mM
173 TrisHCl, 20mM EDTA, CTAB 2% w/v) to each tube which were incubated in an Aosheng
174 MSC-100 shaking incubator at 60°C at 400 rpm until DNA/CTAB complexes were visible. After
175 centrifugation, the DNA formed a pellet which was twice cleaned with 70% ethanol by repetitive
176 inversion. The DNA pellet was air-dried to remove residual ethanol and subsequently
177 resuspended in sterile lab-grade water. Extracted DNA was quality checked using NanoDrop,
178 Qubit assay and gel electrophoresis.

179 For single nucleotide polymorphism (SNP) genotyping, 20 µL of extracted DNA was sent to
180 Diversity Arrays Technology Pty Ltd (Canberra, Australia) (DART). The DART organisation
181 provides a process pipeline of whole-genome profiling, without the need for a reference genome
182 (Jaccoud et al. 2001). High-throughput DARTseq technology was used to genotype *Turbo*
183 *militaris* DNA. Here, the PstI-based complexity reduction method (Wenzl et al. 2004) was
184 applied for the enrichment of genomic representation with single copy sequences. This method
185 involved the digestion of DNA samples with a cutting enzyme PstI, paired with a set of
186 secondary frequently cutting restriction endonucleases, ligation with site-specific adapters, and
187 amplification of adapter-ligated fragments. Post digestion with a restriction enzyme pair, a PstI-
188 overhang-compatible oligonucleotide adapter was ligated, and the adapter-ligated fragments

189 were amplified in adherence to standard protocol (Wenzl et al, 2004). To develop SNPs, the
190 DArTseq technology was optimized using two PstI-compatible adapters corresponding to two
191 different restriction enzyme overhangs. The genomic representations were generated following
192 the procedures described by Kilian et al. (2012). Next-generation sequencing technology was
193 implemented using HiSeq2000 (Illumina, USA) to detect SNP markers. Sequence data was
194 analysed using DarTsoft14 and DArTdb (Killian et al. 2012).

195 *SNP calling*

196 In total, 208 individual *T. militaris* were initially genotyped with the DArTseq™ platform
197 yielding a total of 19,837 SNP loci with a mean read depth of 15.31 and 10.73% missing data. To
198 improve SNP quality, while optimising the number of loci available for population genomic
199 analyses, quality control filters, based on the descriptive statistics from the DArTseq™ pipeline,
200 were applied to data using the R package dartR v.2.7.2 (Gruber et al. 2019; Mijangos et al.
201 2022). Prior to SNP quality control, we checked for the presence of genetically related
202 individuals, as their inclusion can lead to biased genetic estimations of downstream analyses. We
203 calculated a similarity genetic distance matrix for individuals on the proportion of shared alleles
204 per pairs of individuals with the function *gl.propShared* in *dartR*. Two individuals, one from
205 Nambucca Heads (NAM07) and one from South Solitary Island (SSI16) were characterised as
206 closely related and were removed from downstream analyses.

207 Two SNP datasets were generated for analyses: dataset ‘1’ for the analysis of overall genetic
208 structure, and dataset ‘2’ for tests of GEAs and the identification of candidate loci. Both datasets
209 were generated by retaining a single SNP per tag, removing secondaries, applying a locus and
210 individual call rate of 80%, reproducibility threshold of 80%, and a Hamming distance threshold
211 of 0.2 to control for the influence of linkage disequilibrium between loci. SNPs were called for
212 dataset ‘1’ by applying a minor allele frequency (MAF) threshold of 0.03 and removing all loci
213 departing from Hardy–Weinberg expectations. In contrast, SNPs were called for dataset ‘2’ by
214 setting MAF to 0.01, and not filtering out SNP loci deviating from by Hardy–Weinberg
215 expectations (all loci included). Finally, we also used *poppr* (Kamvar, Tabima, and Grunwald
216 2014) in the R package to calculate the number of private alleles found in each population, and
217 remove these SNPs using *gl.drop.loc* in package *dartR*. After filtering, a total of 3,527 and 6,852
218 SNP loci for 206 individuals (Table 2) were retained for data sets ‘1’ and ‘2’, respectively.

219 *Tests for population differentiation*

220 Several estimates of genetic diversity were generated using the *poppr* package in R, including
221 observed (HO) and expected (HE) heterozygosity and allelic richness (AR) (Joop Ouborg,
222 Angeloni, and Vergeer 2010). For AR, allele counts were rarefied by the minimum number of
223 individuals genotyped using the *allelic.richness* command in the R package *hierfstat* (Weir and
224 Goudet 2017). Statistical differences in genetic diversity measures among sites was estimated
225 using the *Hs.test* function in the R package *adegenet* (Jombart 2008). Departures from random
226 mating were calculated using F_{IS} (inbreeding coefficient) for the overall dataset and for each
227 sample location using the *basic.stats* function in *hierfstat*.

228 Tests for population genetic structure were subsequently performed, by calculating global
229 population differentiation (F_{ST}) with 95% confidence limits (Weir & Cockerham, 1984), and
230 population pairwise measures of F_{ST} with significance determined using permutation (999) in the
231 *dartR*. Multiple testing effects were corrected for using the Benjamini–Hochberg FDR procedure
232 (Benjamini and Hochberg 1995). An analysis of molecular variation (AMOVA) was performed
233 in the R package *poppr* (Kamvar, Tabima, Everhart et al, 2014), using a model that partitioned
234 variation among sample sites and within sample sites, with significance based on a
235 randomization test with 999 permutations. A mantel test of isolation by distance (IBD) was
236 carried out using the *gl.ibd* (Rousset 1997) function in the R package *dartR* (Gruber et al. 2019)
237 with results visualised in a scatterplot. A discriminant analysis of principal components (DAPC)
238 was performed using *adegenet* package in R (Jombart 2008). The *find.clusters* function was used
239 to detect the number of clusters in the population. The best number of subpopulations has the
240 lowest associated Bayesian Information Criterion (BIC). A cross validation function (*Xval.dapc*)
241 was used to confirm the correct number of PCs to be retained. Finally, we used sparse non-
242 negative matrix factorisation (sNMF) implemented in the R package *LEA* (Frichot and François
243 2015) in R. This algorithm estimates the genetic ancestry components for each sample. For this
244 study, 15 runs were performed with $\alpha = 100$ for each K value (one to eight). The selection of the
245 best number of putative ancestral populations was guided by the cross-entropy criterion
246 (wherein, for K, a plot of the cross-entropy curve formed a ‘knee’) and the results from the best
247 run were visualised using the *barplots* function.

248 *Environmental variables*

249 Physical oceanographic data were downloaded from the Copernicus Marine Environment
250 Monitoring Service (<https://marine.copernicus.eu>), using a 20-year historical time-series
251 encompassing 2001–2020 (daily temporal resolution) matched to each sampling location. These
252 variables included temperature, water flow and productivity: sea surface temperature (SST) at
253 0.05° spatial resolution; remotely-sensed eddy kinetic energy (EKE) at 0.25° spatial resolution;
254 and remotely-sensed chlorophyll a concentration (CHLa) at 0.04° spatial resolution (Table 2).
255 The native spatial resolutions of oceanographic variables were used when matching daily data to
256 sampling locations. Mean values were calculated for each variable by averaging the daily data. In
257 addition, the lowest and highest SST for each location was extracted, and an absolute
258 temperature range variable (*temp.range*) was calculated by subtracting the minimum from the
259 maximum. A *pairs.panels* scatter plot of matrices was generated in the R package *psych* to
260 confirm a lack of (Pearson's) correlation between variables ($r^2 \leq 0.8$), which were subsequently
261 used in GEA analyses (described below).

262 ***Genome–environment association (GEA) analyses***

263 To detect putative genomic signatures of selection, tests for GEAs were conducted using
264 two complementary models: latent factor mixed models (LFMM2) (Caye et al. 2019), and
265 the Bayesian method available in BayPass V1.01 (Gautier 2015). BayPass and LFMM
266 accounted for potentially confounding allele frequency differences due to population
267 structure in a mixed linear model framework, but in different ways: LFMM estimates GEAs
268 when simultaneously correcting for population structure with latent factors, while BayPass
269 uses a neutral covariance matrix constructed from population allele frequencies. The subset
270 of environmental variables described above were used as predictor variables for both
271 analyses.

272 Genotype-environment associations were explored with Baypass V1.01 (Gautier 2015)
273 under the auxiliary (AUX) covariate model (*-covmcmc* and *-auxmode* flags). The first core
274 model (without the environmental data) was run to estimate a covariance matrix (Ω) of
275 population allele frequencies, which is an approximation of genomic differentiation
276 between populations caused by demographic history. In order to reach convergence and
277 reproducibility of the MCMC estimates, five independent runs, each with a randomly
278 chosen seed were performed using default parameters, except for: pilot runs length of 1000

279 iterations, number of sampled parameter values of 1000, and a burn-in period length of
280 2500 iterations. Secondly, the average of the five covariance matrixes was used as input for
281 the auxiliary covariate model to detect evidence of an association corrected for population
282 structure. Environmental variables were scaled using the “-scalecov” option and the same
283 running parameters as the core model were applied. Finally, the strength of association
284 between genotype and the covariates was assessed by calculating the average of the log-
285 transformed Bayes Factor (BF) in deciban units (dB) for each locus and environmental
286 predictor. Significance was determined following Jeffrey’s criterion for decisive
287 associations ($BF_{IS} \geq 20$) (Jeffreys 1939).

288 Latent factor mixed models (LFMM2) tested for linear relationships between environmental
289 variables and genetic variants with random latent factors using a least-square method.
290 Population structure was inferred by estimating individual ancestry coefficients based on
291 sparse non-negative matrix factorisation (SNMF) method implemented in the *snmf* function
292 in the *R* package *LEA* v3.10.2 (Frichot and François 2015). Ancestry coefficients were
293 determined for 1–8 ancestral populations (K) by generating an entropy criterion that
294 evaluates the fit of the statistical model to the data using a cross-validation technique
295 (Frichot and François 2015). The K with the lowest cross-entropy value using 100
296 repetitions for each K value was selected. Subsequently, the optimal factor, $K=1$, was used
297 to inform the LFMM to identify whether allele frequencies were correlated with any of the
298 environmental variables. To increase the statistical power of associations, missing genotype
299 data were imputed via the ‘*impute*’ function in the *LEA* package, using the most common
300 allele frequency observed in each K with the method ‘*mode*’. Next, we used the function
301 *lfmm_ridge* to compute a regularised least-squares estimate using a ridge penalty. Individual
302 associations between each SNP frequency and each environmental variable were assessed
303 using statistics test calibrated using genomic inflation factor (function *lfmm_test*).
304 Corrections for multiple comparisons were applied with the Benjamini-Hochberg algorithm
305 with a false discovery rate (FDR) threshold of 5% (Benjamini and Hochberg 1995).
306 Significance associations were determined using a threshold of 0.001, since the probability
307 of finding a false positive result increases with lower thresholds (Ahrens, Byrne, and Rymer
308 2019).

309 The gradient forest (GF) algorithm was subsequently used to describe the strength of the

310 associations of spatial, environmental variables and candidate loci and to map spatial
311 patterns of allelic turnover in climate space (Ellis, Smith, and Pitcher 2012; Fitzpatrick and
312 Keller 2015). Gradient forest is a machine learning method initially developed to model the
313 turnover of ecological community assemblages in relation to environmental gradients (Ellis
314 et al. 2012). Recently, this method has been adapted as a landscape genomics toolbox,
315 substituting allele frequencies at genetic loci for species to model allelic turnover in climatic
316 space (Fitzpatrick and Keller 2015). The turnover functions in gradient forest allow for
317 inference of the environmental predictors driving observed changes in allele frequency
318 (Fitzpatrick and Keller 2015). Analyses were implemented in the *R* package *gradientForest*
319 (Ellis et al. 2012), using a regression tree-based approach to fit a model of responses
320 between genomic data and environmental variables (Capblancq et al. 2020). Specifically,
321 adaptive genetic variation turnover were modelled on the seascape using the candidate
322 SNPs (derived from LFMM, and BayPass) set as the response variables. The machine
323 learning algorithm partitioned allele frequencies at numerous splits values along each
324 environmental gradient and calculated the change in allele frequencies for each split. The
325 split importance (i.e., the amount of genomic variation explained by each split value) was
326 cumulatively summed along the environmental gradient and aggregated across alleles to
327 build a non-linear turnover function to identify loci that are significantly influenced by the
328 predictor variable (Ellis et al. 2012). The analysis was run over 500 regression trees for each
329 of the four environmental variables with all other parameters at default settings. The
330 cumulative goodness-of-fit among SNPs was represented as an R^2 value indicating how well
331 a predictor explained changes in allele frequency and which predictors were most important
332 in predicting genomic changes. The resulting multidimensional genomic patterns were
333 summarised using principal component analysis (PCA), allowing the relative importance of
334 predictor variables on allelic turnover to be visualised. Finally, using the top gradient forest
335 model, we interpolated genetic composition and allelic turnover across the sampling range
336 of eastern Australia.

337 **Results**

338 *Overall population genetic structure*

339 Patterns of genetic diversity did not differ greatly across the eight sample locations ($H_O = 0.186$
340 $- 0.212$; $AR = 1.630 - 1.640$; Table 3). Most sites showed a weak excess of heterozygotes ($F_{IS} =$
341 $0.048 - 0.118$), however these estimates did not differ significantly from zero ($p > 0.01$; Table
342 3). Overall genetic differentiation was found to be significant, but weak and close to zero (global
343 $F_{ST} = 0.002$, $p < 0.001$) indicating a lack of genetic structure among sampling locations (Table
344 3). These findings are further supported by weak, yet significant, estimates of genetic
345 differentiation among all pairs of sampling locations ($F_{ST} = 0.000 - 0.004$, Table 4). AMOVA
346 also indicated a lack of overall genetic structure indicating genetic variance attributed to
347 differences among sites to be non-significant (0.212 , $p > 0.05$) while the majority of variance
348 was explained by genetic variation between individuals within sites (99.86 , $p > 0.05$). Similarly,
349 DAPC and LEA analyses indicated a lack of genetic structure, both identifying a single
350 population cluster ($K = 1$). Finally, Mantel tests revealed no significant relationship between
351 genetic differentiation and distance between sampled locations ($R^2 = 0.03356$, $p = 0.278$)
352 providing further evidence of panmixia.

353 *Genotype x environment associations*

354 BayPass and LFMM each identified a number of candidate SNP loci exhibiting significant
355 genotype-by-environment associations for each of the environmental predictors tested (Fig. 2.
356 BayPass detected between 0 and 2 SNPs with significant correlations ($\log_{10}(BF) > 20$) for each
357 of the environmental predictors, with zero overlap in candidates between predictors. In contrast,
358 LFMM detected between 5 and 12 SNPs that were significantly correlated with each of the
359 environmental predictors (Fig. 2), but with only 2 loci overlapping between the EKE and SST
360 predictor variables. Concordance of candidate SNP loci between methods was low (3 loci only;
361 Figure 2) but expected given these methods have varying sensitivities to detecting loci under
362 selection, use different methods for controlling for demography, and adopt different association
363 algorithms. Given that *T. militaris* exhibits panmixia, its lack of population structure was likely
364 to influence the inference of omega matrix and K clusters on BayPass and LFMM respectively.
365 Furthermore, LFMM tests for relationships between individual-based allele frequencies whereas
366 BayPass is at population level. Additionally, sampling design is likely to have had an influence,
367 wherein a strategic sampling design accommodated to environmental heterogeneity and spatial
368 variation on a landscape is essential to potentially identify and validate patterns of local
369 adaptations across natural populations. Overall, GEAs indicated that a larger proportion of SNPs

370 were significantly associated with temp range (13) and CHLa (11), followed by EKE (7) and
371 SST (5).

372 Gradient Forest modelling used the unique candidate adaptive loci detected by both LFMM and
373 BayPass (Lu et al. 2021), with 39 in total, 4 in BayPass and 35 in LFMM, but three were shared
374 between both analyses and one was shared within LFMM (variables EKE and SST) (thus $39 - 3 - 1 = 35$
375 unique candidate loci). Gradient forest analyses showed 14 of the 35 candidate SNP loci
376 to be significantly correlated with environment (R^2 values > 0 ; mean = 0.08, range 0.003 – 0.3).
377 Overall, EKE, CHLa and SST were found to be the most important predictors of genomic
378 variation, while temp range had less of an effect (Figure 3). Turnover functions from the GF
379 model show the weighted cumulative importance values and sharp turnovers for all
380 environmental predictors, but again with temp range having lower importance relative to all
381 other predictors (Figure 3a). Biplots based on the first two principal components captured
382 approximately 99% of the total variation and point to EKE and SST as the most prominent
383 drivers of genomic variation (Figure 3b). A spatial depiction of genomic composition in multi-
384 dimensional climatic space based on PCA is provided in the allelic turnover map (Figure 3c).
385 The map indicates that the turnover of putatively adaptive allelic variation tracks closely with
386 latitude, with the genomic composition of northern and southern most sampling locations being
387 distinct from those from geographically intermediate locations (Fig. 3d).

388 Discussion

389 Understanding spatial patterns of gene flow and local adaptation can help predict species
390 responses to climate change and to identify populations most at risk of maladaptation (Hoffmann
391 and Sgrò 2011; Sexton et al. 2009). Such information is critically important for assisting with the
392 adaptive management of commercially important marine species, many of which are already
393 showing signs of climate stress (Cheung et al. 2013; Pinsky et al. 2018; Sunday et al. 2015). This
394 study represents the first population genomic analysis of *T. militaris*, a commercially and
395 culturally important trochid marine gastropod from the east coast of Australia, with the purpose
396 of informing fisheries managers about vulnerability of this species to future climate change.
397 Analyses of SNP genotypes across the species' entire distribution spanning seven degrees of
398 latitude and 3.9°C in mean annual sea surface temperature indicate the presence of a single
399 admixed, and potentially panmictic, demographic unit with no evidence of genetic subdivision

400 along the entirety of its range. Furthermore, significant genotype associations with
401 heterogeneous habitat features were observed at regional spatial scales, including associations
402 with sea surface temperature, ocean currents, and nutrients, indicating possible adaptive genetic
403 differentiation among sample locations. Combined, these findings provide insights into the
404 potential resilience of *T. militaris* to changing marine climates and the potential influence of
405 gene flow and selection on future adaptive responses.

406 *Evidence of panmixia*

407 Our analyses point to a lack of genetic structure across the entire distribution of *T. militaris*
408 indicating widespread gene flow along the eastern seaboard of Australia. Such genetic patterns
409 are also found amongst other trochoid taxa with long pelagic larval phases which are expected to
410 facilitate long distance dispersal (Berry et al. 2019; Díaz-Ferguson et al. 2010; Nikula et al.
411 2011; Silliman, Grosholz, and Bertness 2009). Several other eastern Australian marine
412 invertebrates also exhibit high gene flow facilitated by larval traits including the Crown-of-
413 thorns sea star (*Acanthaster* spp.) (Pratchett et al. 2015), the surf bivalve *Donax deltoides*
414 Lamarck, 1818 (Murray-Jones and Ayre 1997; Miller et al. 2013) and the black sea-cucumber
415 *Holothuria (Mertensiothuria) leucospilota* (Brandt, 1835) (Chieu et al. 2023). Here gene flow
416 and population structure has been linked to the long-distance dispersal of pelagic larvae
417 facilitated by a fast-flowing east Australian current (EAC). While knowledge of the reproductive
418 biology or larval competency of *T. militaris* is poor, our results suggest that the species may also
419 generate long-lived, planktotrophic larvae contributing to high levels of biological connectivity
420 across its distribution (Cowen and Sponaugle 2009). Overall, these findings are consistent with
421 previous genetic studies on trochoid taxa indicating population admixture over vast geographical
422 areas. In this case we have provided evidence of gene flow among potentially locally adapted
423 populations spanning major environmental gradients.

424 *Evidence of local adaptation*

425 Despite a lack of overall genetic structure across the sampling distribution, significant genotype
426 associations with heterogeneous habitat features were observed across the sampling distribution,
427 including associations with annual mean sea surface temperature, sea surface temperature range,
428 EKE and productivity (CHLa). Drift processes leading to neutral genetic structure are often
429 suppressed in broadcast spawning marine organisms with large population sizes (Gélin et al.

2017; Palumbi 2003; Pinsky and Palumbi 2014), but numerous studies have shown that adaptive genetic divergences can still be established and maintained under strong selection pressure (Hendry 2017; Nosil 2012; Schluter 2000). In fact, many studies have demonstrated that adaptive variation can be maintained despite high levels of gene flow specifically in marine invertebrates, including gastropod snails (Miller et al. 2019; Sandoval-Castillo et al. 2018). However, these findings are based on correlative tests only and should be interpreted with caution, as controlled mechanistic experiments are needed to validate these patterns and drivers of putative adaptive differentiation (Savolainen, Lascoux and Merilä 2013; Stinchcombe and Hoekstra 2008). Also, while our sampling regime was designed to correct for geographical distance, we cannot rule out the possible influence of artefactual associations (i.e., false positives) and SNP associations with other environmental factors varying with latitude (such as light intensity, dissolved oxygen levels and rainfall-driven variation in salinity). Nevertheless, our findings are consistent with many studies of marine species which demonstrated genetically determined clines related to climatic variables in Australia and overseas (Poloczanska et al. 2013, 2016; Wernberg et al. 2016).

Resilience to future climatic challenges

The coastal waters of south-eastern Australia are a recognised climate change hotspot with warming occurring above the global average with changes enhanced by strengthening of the EAC (Cresswell, Peterson, and Pender 2016; Malcolm et al. 2011; Oliver et al. 2017). Predictions anticipate warming to continue to increase, generating deteriorating marine conditions for sessile taxa that are already at their thermal limits (e.g. Davis et al. 2021). Within the EAC, the effect of warming and changes to circulation may also counter one another, with warming enhancing larval survival, but a strengthened current reducing larval supply to the coast by restricting cross-current larval dispersal (e.g. in lobster, Cetina-Heredia et al, 2015). Our findings suggest that *T. militaris* may have the capacity to adapt to future climatic challenges, assisted by both widespread gene flow across environmental gradients and the availability of standing genomic variation for selection to act upon. Findings of putative adaptive genetic differentiation associated with temperature, nutrients and ocean currents is a particularly important finding, suggesting that standing genetic variation may be available for selection to act on to counter future environmental change, assisted by widespread gene flow, high fecundity (Romolo and Trijoko 2021) and highly probable short generation time in this species (Kimani 1996; Romolo and Trijoko 2021).

461 While our findings suggest that *T. militaris* is likely to be generally resilient to shifts in the
462 physical ocean climate, aggregations in some areas may still be vulnerable to risks of
463 maladaptation. Those most at risk are likely to include locally adapted populations, where
464 projected local changes in climate are high, and connections to non-local aggregations are
465 relatively weak (Hoffmann and Sgro 2011). Recent climate projections indicate that many low
466 energy embayment habitats are likely to experience greater increases in SST than open coastal
467 habitats (Guyondet et al. 2015; Scanes, Scanes, and Ross 2020; Vila-Concejo et al. 2007); Also,
468 biophysical models suggest that the biological connections between low energy embayment and
469 high energy open coastal habitats can be weak in some marine invertebrates from south-eastern
470 Australia (Riginos et al. 2016; Treml et al. 2015). Consequently, it is possible that locally
471 adapted aggregations from low energy embayment habitats may be most vulnerable to climate
472 change effects, where gene flow is unlikely to assist local aggregations in adapting to warming
473 sea surface temperatures via the migration of thermally adapted genotypes. In such cases
474 adaptive management strategies might be needed, including the assisted migration of thermally
475 adapted genotypes to populations showing signs of climate stress. Such approaches are being
476 widely advocated as a tool for “climate proofing” threatened marine and terrestrial animal and
477 plant communities (Aitken & Whitlock, 2013; Prober et al., 2015; Layton et al., 2020;
478 Hoffmann, Miller and Weeks 2021). Although the northernmost Hastings Point population is not
479 genetically isolated, its position at the northern trailing edge may render it vulnerable to
480 stochastic events, such as heatwaves (Ab Lah et al, 2018; Mamo et al, 2019). Vulnerability in
481 trailing edge populations can be amplified by genetic impoverishment through loss of
482 individuals, without replenishment through immigration (gene-flow), consequently exposing
483 these populations to the risk of localised extinction (Clark et al. 2020; Coleman et al. 2020).
484 Indeed, this species has undergone prior range shifts indicating that the trailing (warm) edge is
485 likely to be vulnerable to ongoing warming.

486 *Implications for fisheries management*

487 Sustainable fisheries management requires information on factors that influence the resilience of
488 individual fishing stocks to fishing pressure and environmental disturbance (Astles et al. 2006;
489 Kenny et al. 2018). This includes understanding the geographic boundaries of biological
490 populations and the recruitment potential of individual stocks persisting within and across these
491 populations (Binks et al. 2019; Roughgarden, Iwasa, and Baxter 1985) and how this might be

492 altered under climate change. In the case of *T. militaris*, the presence of a single panmictic
493 population unit across its distributional range, generally means that the opportunity for
494 recolonisation following depletion events (overharvesting or environmental disturbance) are
495 enhanced for central and southern sub-populations. However, for the northern trailing edge, the
496 opportunity for repopulation with genotypes from pools of genetic diversity further south may be
497 hampered by the dominant poleward flow of the EAC. Furthermore, increasing human
498 population and harvesting pressure may reduce local abundance (Cooling and Smith 2015) and
499 the opportunity for thermal adaption and repopulation.

500 **Conclusions**

501 Knowledge of population genomics, particularly adaptive structure, is important for fisheries
502 management and can be used to estimate vulnerability and adaptability of stocks under climate
503 change. This study revealed that the harvested gastropod, *T. militaris*, is panmictic across its
504 distributional range with little variation in genetic diversity and can be considered as a single
505 stock. As such, it has the genetic capacity to survive and proliferate within its environmental
506 niche and is likely to continue to track ocean temperatures by shifting its entire distribution
507 poleward. Genomic studies can improve management of harvested species under climate change
508 by providing insights into adaptive capacity and help identify opportunities for strategic adaptive
509 management (van Oppen & Coleman 2022).

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515

516

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Table 1 (on next page)

Predictor variables sourced from Copernicus Marine Environment Monitoring Service used in genotype by environmental association analyses.

- 1 **Table 1.** Details of samples, sampling location and location based environmental covariables for *Turbo militaris* in New South Wales. (*N*=number, SST = mean
 2 sea surface temperature, Temp range = total temperature variance, EKE = mean kinetic energy, CHL*a* = mean chlorophyll *a*).

Location	Latitude	Longitude	Date collected	<i>N</i> samples	SST (°C)	Temp range (°C)	EKE	CHL <i>a</i>
Hastings Point	28° 21' 38.09" S	153° 34 '44.21"E	30 Mar 2021	25	23.35	8.91	0.1391	2.1094
Woody Head	29° 21 '47.70" S	153° 22' 25.96"E	08 Apr 2021	25	22.34	9.52	0.0632	3.8472
South Solitary Island	30° 12' 19.31" S	153° 16' 01.24 "E	12 May 2021	25	22.42	10.17	0.1204	0.9517
Split Solitary Island	30° 14' 24.07"S	153° 10' 50.71"E	12 May 2021	30	22.37	10.14	0.0741	2.0323
Nambucca Heads	30°38'47.49"S	153°01'15.10"E	26 Mar 2021	23	22.23	10.23	0.1223	2.0319
Crowdy Head	31° 50' 16.91"S	152° 45 '04.54"E	28 Apr 2021	22	21.34	9.94	0.0820	2.7083
Newcastle	32° 57' 31.70"S	151° 45' 02.72"E	12 Apr 2021	31	20.39	9.12	0.0178	2.2840
Jervis Bay (Plantation Pt)	35° 06 '16.15"S	150°41'53.70"E	20 Aug 2021	25	19.46	10.24	0.0311	2.9248
Total				206				

3

Table 2 (on next page)

Details of samples

Table . Details of samples, sampling location and location based environmental covariables for *Turbo militaris* in New South Wales. (N=number, \bar{X} SST = mean sea surface temperature, temp range = total temperature variance, \bar{X} EKE = mean kinetic energy, \bar{X} CHLa mean chlorophyll a).

1 **Table 2.** Predictor variables sourced from Copernicus Marine Environment Monitoring Service used in genotype by
2 environmental association analyses.

Predictor	Description	Spatial Resolution	Units
SST	Daily global sea surface temperature reprocessed (level 4) from Operational SST and Ice Analysis system downloaded from CMEMS (product #010_011).	0.05°	°C
CHLa	8-day composite mass concentration of chlorophyll <i>a</i> in seawater (level 4) from Globcolour downloaded from CMEMS (product #009_082).	4 km	mg m ⁻²
EKE	Daily eddy kinetic energy computed from zonal and meridional velocity components from the Sea Level Thematic Assembly Centre downloaded from CMEMS (product #008_047).	0.25°	m ² s ⁻²
Year	Calendar year (incorporated as a random intercept term in mixed models).	-	-

3

Table 3 (on next page)

Summary of descriptive statistics

Table . Summary of descriptive statistics. Sample size (n), mean allelic richness (AR), and genetic diversity indices including expected (HE) and observed (HO) heterozygosity, and inbreeding coefficients (FIS) at each site, based on the complete filtered dataset (n = 207 individuals).

1 **Table 3.** Summary of descriptive statistics. Sample size (n), mean allelic richness (A_R), and genetic diversity indices
2 including expected (H_E) and observed (H_O) heterozygosity, and inbreeding coefficients (F_{IS}) at each site, based on
3 the complete filtered dataset ($n = 207$ individuals).

Location	n	A_R	H_E	H_O	F_{IS}
Hastings Point	25	1.635	0.218	0.190	0.110
Woody Head	25	1.633	0.216	0.189	0.107
Split Solitary Is	30	1.632	0.217	0.189	0.111
South Solitary Is	25	1.638	0.219	0.202	0.076
Nambucca Heads	23	1.640	0.221	0.212	0.048
Crowdy Head	22	1.633	0.217	0.188	0.111
Newcastle	31	1.630	0.215	0.186	0.118
Jervis Bay	25	1.630	0.217	0.194	0.096

4

Table 4(on next page)

Pairwise estimates of pairwise F_{ST} among sample locations. Bolded F_{ST} values were found to be significant after multiple corrections ($p < 0.05$).

- 1 **Table 4.** Pairwise estimates of F_{ST} among sample locations. Bolded F_{ST} values were found to be significant after
 2 multiple corrections ($p < 0.05$). Numbers in lower diagonal are p values

	Nambucca Heads	Woody Head	Newcastle	Hastings Point	South Solitary Is	Crowdy Head	Split Solitary Is	Jervis Bay
Nambucca Heads		0.0037	0.0037	0.0031	0.0033	0.0028	0.0028	0.0043
Woody Head	0.00		0.0022	0.0019	0.0034	0.0029	0.0027	0.0038
Newcastle	0.00	0.00		0.0017	0.0024	0.0016	0.0014	0.0021
Hastings Point	0.00	0.00	0.01		0.0018	0.0007	0.0014	0.0017
South Solitary Is.	0.00	0.00	0.00	0.00		0.0016	0.0020	0.0031
Crowdy Head	0.00	0.00	0.00	0.11	0.00		-0.0002	0.0016
Split Solitary Is	0.00	0.00	0.00	0.00	0.00	0.69		0.0020
Jervis Bay	0.00	0.00	0.00	0.01	0.00	0.00	0.00	

3

Figure 1

Map of *Turbo militaris* collection sites from the eastern seaboard of Australia

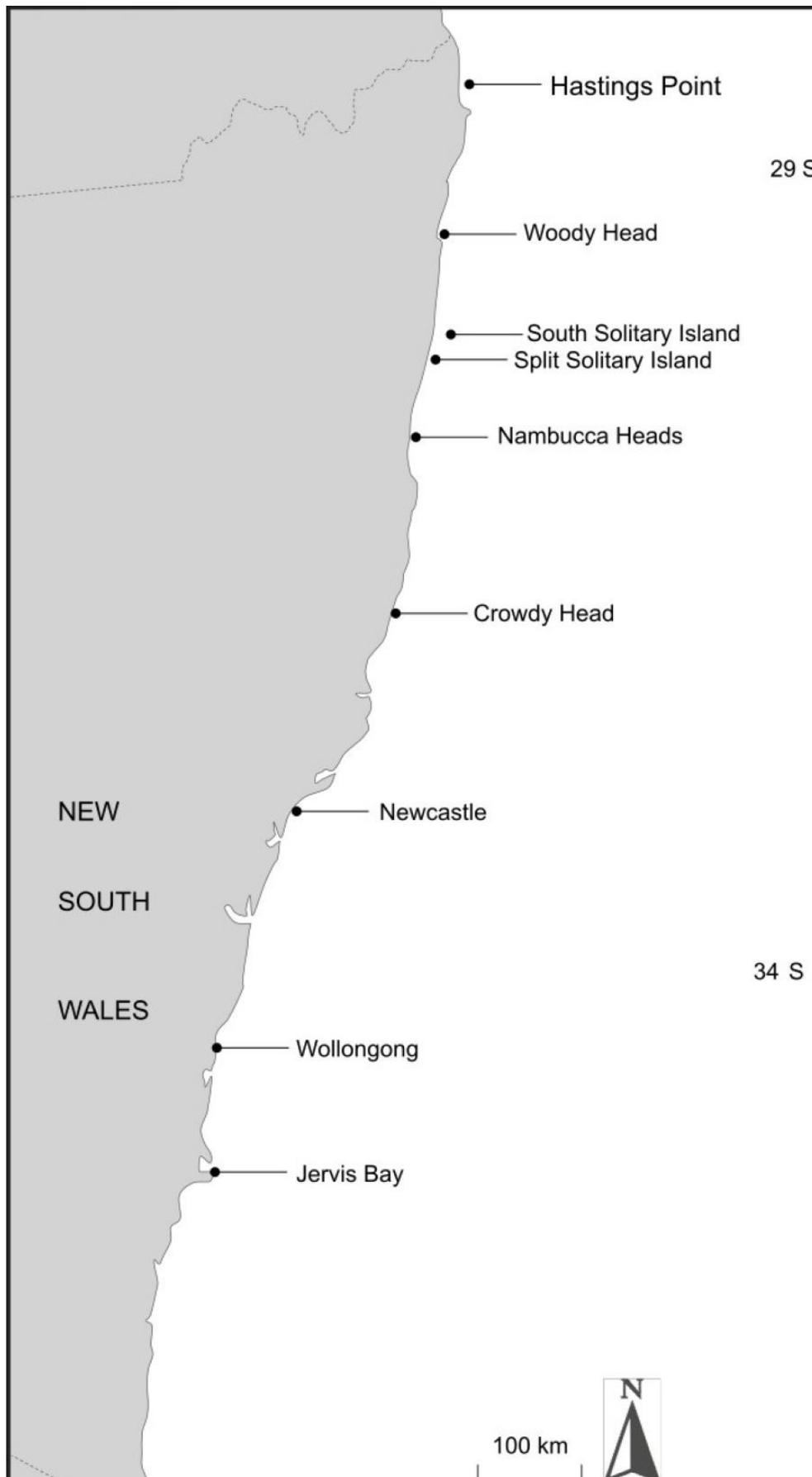


Figure 2

Results of genotype x environment association analyses show the strength of associations between individual loci and environmental variables identified by BayPass and LFMM2

Bar plots show the number of putative candidate loci associated with each environmental variable.

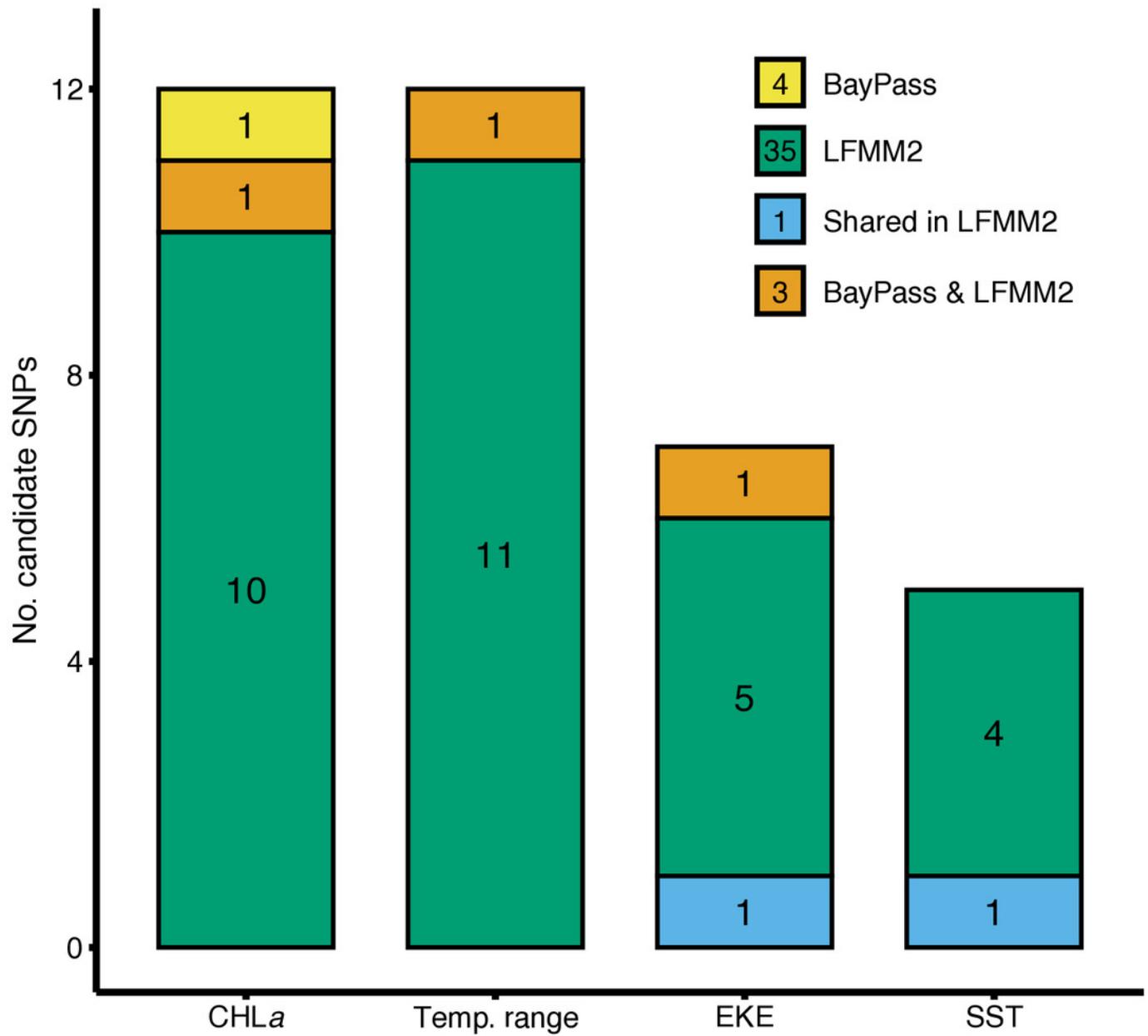


Figure 3

Gradient Forest outputs

(a) Overall relative importance for each environmental variable describing the turnover in allele frequencies from the Gradient Forest model; **(b)** cumulative importance curves showing overall pattern of genomic compositional change (R^2 , y-axis) for each environmental gradient (x-axis). Turnover functions for each curve are aggregated across all candidate loci. The curve shape indicates the rate of change in allele frequencies along the environmental gradient, and the maximum height indicates the total turnover in allele frequencies. Predictor variables shown here include: chlorophyll *a* concentration (CHLa), eddy kinetic energy (EKE), sea surface temperature (SST) and sea surface temperature range (temp.range); **(c)** Influence of environmental variation on candidate SNP allele frequencies inferred from Gradient Forest analyses as a PCA plot illustrating the influence of environmental variation on allele frequencies for candidate loci. Background colours denotes environmental space, whereas black dots represent the principal component (PC) scores associated with the sampling locations; **(d)** Predicted spatial turnover in allele frequencies interpolated across the study area, with black dots as labelled sampling locations. Similar colours indicated areas expected to have similar genetic composition, while divergent colours indicate divergent putatively adaptive genotypes. Colours are based on the first three principal components of transformed environmental variables.

