

Differential relationships between habitat fragmentation and within-population genetic diversity
of three forest-dwelling birds

Abridged title: Habitat fragmentation and population genetic variation

Benjamin Zuckerberg^{1,2}, Matthew D. Carling^{2,3}, Roi Dor^{2,4}, Elise D. Ferree^{2,5}, Garth M. Spellman⁶, and Andrea K. Townsend^{2,7}

8

¹Department of Forest and Wildlife Ecology, University of Wisconsin, 226 Russell Labs,
Madison, WI 53706, USA, bzuckerberg@wisc.edu

²Cornell Lab of Ornithology, Cornell University, Ithaca, NY 14850, USA

³Department of Zoology and Physiology, Berry Biodiversity Conservation Center, University of
Wyoming, Laramie, WY, 82071, USA, mcarling@uwyo.edu

⁴Department of Zoology, Tel Aviv University, Tel Aviv 69978, Israel, roidor@post.tau.ac.il

⁵Keck Science Department, Claremont Colleges, Claremont, CA, 91711 USA,
EFerree@kecksci.claremont.edu

⁶Department of Biology, Black Hills State University, Spearfish, SD 57799, USA,
Garth.Spellman@bhsu.edu

⁷Department of Wildlife, Fish, and Conservation Biology, University of California, Davis
One Shields Avenue, Davis, CA 95616, USA, aktownsend@ucdavis.edu

22

Abstract Habitat fragmentation is a major driver of environmental change affecting
24 wildlife populations across multiple levels of biological diversity. Much of the recent research in
landscape genetics has focused on quantifying the influence of fragmentation on genetic
26 variation among populations, but questions remain as to how habitat loss and configuration
influences within-population genetic diversity. Habitat loss and fragmentation might lead to
28 decreases in genetic diversity within populations, which might have implications for population
persistence over multiple generations. We used genetic data collected from populations of three
30 species occupying forested landscapes across a broad geographic region: Mountain Chickadee
(*Poecile gambeli*; 22 populations), White-breasted Nuthatch (*Sitta carolinensis*; 13 populations)
32 and Pygmy Nuthatch (*Sitta pygmaea*; 19 populations) to quantify patterns of haplotype and
nucleotide diversity across a range of forest fragmentation. We predicted that fragmentation
34 effects on genetic diversity would vary depending on dispersal capabilities and habitat specificity
of the species. Forest aggregation and the variability in forest patch area were the two strongest
36 landscape predictors of genetic diversity. We found higher haplotype diversity in populations of
P. gambeli and *S. carolinensis* inhabiting landscapes characterized by lower levels of forest
38 fragmentation. Conversely, *S. pygmaea* demonstrated the opposite pattern of higher genetic
diversity in fragmented landscapes. For two of the three species, we found support for the
40 prediction that highly fragmented landscapes sustain genetically less diverse populations. We
suggest, however, that future studies should focus on species of varying life-history traits
42 inhabiting independent landscapes to better understand how habitat fragmentation influences
within-population genetic diversity.

44 *Key words:* haplotype diversity, mitochondrial DNA, nucleotide diversity, *Poecile*
gambeli, *Sitta carolinensis*, *Sitta pygmaea*

46 INTRODUCTION

Ecologists often study how habitat composition and configuration influences the abundance and
48 occurrence of organisms (Turner et al. 2001, Fahrig 2003), but in recent years, the field of
landscape genetics has focused on how these landscape features influence genetic variation
50 (Anderson et al. 2010, Sork and Waits 2010, Storfer et al. 2010). Landscape genetics studies
have successfully incorporated the role of isolation and geographic distance in explaining genetic
52 variation among populations (e.g., Jenkins et al. 2010); however, there has been a persistent
interest in examining how habitat composition and configuration influences genetic diversity
54 within populations (Bruggeman et al. 2010, Sork and Waits 2010, Thomassen et al. 2010). This
interest has been driven, in part, by the concern that habitat fragmentation might lead to
56 decreases in genetic diversity within animal populations, which might have strong implications
for the persistence of populations over multiple generations (Frankham et al. 2010).

58 The process of habitat loss and fragmentation produces landscapes characterized by
isolated habitat patches of decreasing size and continuity and has important implications on
60 genetic diversity. Landscapes characterized by habitat loss and fragmentation are thought to
support smaller effective population sizes (Wilcove 1985, Debinski and Holt 2000, Chalfoun et
62 al. 2002), which neutral theory predicts will lead to decreased genetic diversity (Hartl and Clark
2007). Empirical evidence seems to support this prediction as birds with relatively larger
64 populations and broader distributions are genetically more diverse (Moller et al. 2008). For many
species, one of the mechanisms driving this pattern is that fragmentation causes decreased rates
66 of individual movement and dispersal among isolated habitat patches (Belisle et al. 2001),
leading to a subdivided and more variable population (Boulinier et al. 1998, Boulinier et al.
68 2001, Donovan and Flather 2002). In turn, random genetic drift and reduced gene flow leads to

an eventual decline of within-population genetic diversity as a result of declining and/or
70 disconnected populations (Templeton et al. 1990, Young et al. 1996, Keyghobadi 2007).

Comparative studies on the genetic impact of habitat fragmentation among populations
72 generally support the prediction of reduced genetic diversity in fragmented landscapes
(Keyghobadi 2007). Many of these studies, however, are plagued by analytical and
74 methodological limitations. First, studies of habitat fragmentation often confound habitat loss
with the process of habitat fragmentation per se (the spatial breaking apart of habitat independent
76 of habitat loss) (Fahrig 2003). A potential reason for this limitation results from a lack of
replication of independent landscapes by which to assess variability in habitat configuration
78 (McGarigal and Cushman 2002). Second, most comparative studies compare genetic structure
within a control and fragmented landscape in the same geographic region. As a result, it is
80 difficult to identify which aspects of fragmentation might be driving changes in genetic
responses and differentiate those from broader-scaled regional patterns (Johansson et al.
82 2005). Third, few studies focus on multiple species with varying life history characteristics. It is
likely that species of different characteristics of dispersal and habitat specificity could mediate
84 the genetic response to fragmentation at a landscape scale (Van Houtan et al. 2007, Bowie 2011,
Callens et al. 2011). Genetic diversity is a population characteristic that should be particularly
86 sensitive to the spatial distribution of habitat, and there is a need to evaluate the characteristics of
habitat loss and fragmentation that could lead to genetic variation among multiple species
88 inhabiting a continuum of fragmentation.

We quantified patterns of within-population genetic diversity for bird populations
90 exposed to a range of habitat fragmentation for three species of passerine birds found throughout
the western United States: Mountain Chickadee (*Poecile gambeli*), White-breasted Nuthatch

92 (*Sitta carolinensis*) and Pygmy Nuthatch (*Sitta pygmaea*). Previous molecular analyses have
suggested that some of the range-wide patterns of genetic structure and among-population
94 diversity in these three species can be explained by historical isolation and subsequent expansion
from glacial refugia in the Pleistocene. In *P. gambeli*, two clades are consistent with isolation in
96 two refugia during glacial advances (Spellman et al. 2007), and among western populations of *S.*
carolinensis, two of three clades decrease in nucleotide diversity with increasing latitude, a
98 pattern consistent with expansion from southern glacial refugia (Spellman and Klicka 2007). For
S. pygmaea, expansion from a single glacial refugium in southern coastal California may have
100 led to a positive correlation of nucleotide diversity with longitude (Spellman and Klicka 2006).
However, these studies primarily focused on phylogeographic patterns of genetic diversity
102 among populations as opposed to genetic characteristics within independent populations.

Our objective was to use existing genetic data to test the relationship between habitat
104 fragmentation and within-population genetic diversity for the above three forest-breeding bird
species across a range of fragmentation. We predicted that the relationship between
106 fragmentation and within-population genetic diversity, as measured by haplotype and nucleotide
diversity, would vary among species according to their dispersal capabilities and habitat
108 requirements. Specifically, we predicted that the genetic diversity of the *P. gambeli*, with its
intermediate dispersal distance (McCallum et al. 1999) and habitat specificity might not vary
110 with fragmentation. We predicted that genetic diversity would be higher for the *S. carolinensis* in
fragmented landscapes, because of their longer dispersal distances (Grubb and Pravosudov 2008)
112 and relatively diverse habitat associations. Finally, we predicted that genetic diversity would
decrease in fragmented landscapes for the *S. pygmaea*, with its relatively poor dispersal
114 capabilities (Kingery and Ghalambor 2001) and strict habitat associations, because it would be

less likely to disperse across a fragmented landscape (Van Houtan et al. 2007). Generalist species
116 that exhibit relatively greater rates of dispersal should be less susceptible to the impacts of
habitat fragmentation and its associated processes that erode local genetic diversity over time
118 (Mech and Hallet 2001, Vandergast et al. 2007, Bowie 2011).

120 METHODS

Study species and genetic sampling

122 The distribution of each of our study species, *P. gambeli*, *S. carolinensis*, and *S. pygmaea*,
matches the distribution of their preferred forest habitats, coniferous forests, pine-oak and long-
124 needle pine, respectively. *Poecile gambeli* generally prefers montane coniferous forests and areas
dominated by pine, spruce-fir (*Picea-Abies*), and piñon-juniper (*Pinus-Juniperus*) (McCallum et
126 al. 1999). They will occur in mixed coniferous-deciduous habitats, but show preferential use of
conifer stands in these settings (McCallum et al. 1999). *Poecile gambeli* generally does not
128 migrate, but most yearlings exhibit natal dispersal, although young dispersers sometimes move
only as far as adjacent family groups (McCallum et al. 1999). *Sitta carolinensis* is more closely
130 associated with mature deciduous woodland, but can also breed in mixed deciduous, coniferous
forest, and a variety of montane woodland habitats (Grubb and Pravosudov 2008). They exhibit
132 somewhat irruptive migration and disperse up to 10 km in their first year, and although little is
known about natal dispersal distances for *S. carolinensis* (Grubb and Pravosudov 2008), mean
134 natal dispersal distances were less than 10 km in both a Belgian population (Matthysen and
Schmidt 1987) and German population (Winkel 1989) of Eurasian nuthatches (*Sitta europaea*).
136 Finally, *S. pygmaea* is the most specific in terms of habitat requirements. *Sitta pygmaea*
demonstrates a strong and almost exclusive preference for long-neededled pine forests and are co-

138 extensive with that of ponderosa pine (*Pinus ponderosa*), Jeffrey pine (*Pinus jeffreyi*), and
similar species (Kingery and Ghalambor 2001). They are also the most sedentary of the three
140 study species, with no migration and very short natal dispersal distances (< 500 m; Kingery and
Ghalambor 2001). Recorded mean natal dispersal distance for *S. pygmaea* centers around 287 m
142 (range = 0-533 m; Norris 1958). Evidence from individual populations also suggests that each
species is most likely to occur in relatively intact forested areas when compared to fragmented
144 landscapes (Brawn and Balda 1988, McCallum et al. 1999, Villard et al. 1999). Because of their
wide distribution across western North America and reliance on forest habitats, these species
146 created a suitable system to examine the influence of habitat fragmentation on within-population
genetic diversity.

148 For each species, we used available sequences of the mitochondrial gene NADH
dehydrogenase subunit 2 (ND2; 1041 bp) from previously published phylogeographic studies on
150 the three species (Spellman and Klicka 2006, 2007, Spellman et al. 2007). We used previously
generated ND2 sequences from 216 *P. gambeli* individuals (22 populations), 92 *S. carolinensis*
152 individuals (13 populations) and 132 *S. pygmaea* individuals (19 populations; Fig. 1; Table 1).
The location of sampling sites emphasized protected areas with limited anthropogenic
154 disturbance (e.g., National Forests, State Parks, and State Game Preserves).

156 *Genetic data analysis*

Using ARLEQUIN v3.11, we estimated two indices of within-population genetic diversity:
158 haplotype diversity and pairwise nucleotide diversity (Excoffier et al. 2005). Haplotype diversity
is the probability that two randomly chosen haplotypes are different in the sample, whereas
160 nucleotide diversity is the average pairwise nucleotide difference between two sequences in each

population. Nucleotide diversity is haplotype diversity at the nucleotide level; therefore, it
162 captures not only how many different haplotypes are in a population, but also how different those
haplotypes are from each other. Low haplotype and nucleotide diversity can indicate low
164 underlying effective population size (Awise 2000), the influence of recent expansions from
glacial refugia (Spellman and Klicka 2006) or cyclical range shifts (Jaarola et al. 1999). These
166 two indices capture different aspects of genetic variation, but they may be statistically correlated.
We performed separate analyses for each of the species including sequences from all available
168 individuals from each population (Table 1).

170 *Landscape analysis*

We defined landscapes as a 10 km buffer around each sampled population's centroid to represent
172 the spatial extent encountered by most dispersing individuals within a single generation.
Although natal dispersal distances have not been fully characterized for our focal species,
174 available evidence suggests that these maximum dispersal distances are within a 10 km-radius
landscape (see *Study species and genetic sampling*).

176 We analyzed the 2001 National Land Cover Data (NLCD) to characterize the cover and
configuration of forest cover in the 10 km-radius landscape. The 2001 NLCD consists of 16 land
178 cover classes modeled over the conterminous United States at a 30 m cell resolution and a 0.40
ha minimum mapping unit (Homer et al. 2007). We calculated the proportion of each land cover
180 type within each landscape, but because the landscapes tended to be dominated by two primary
cover types (deciduous or coniferous forest and early-successional cover types), we chose to
182 aggregate the land cover classes to represent a binary habitat cover map of forest (including
deciduous/mixed forest or evergreen forest types) and non-forest (including early-successional)

184 cover types. Although there are many metrics available for quantifying landscape heterogeneity
(Cushman et al. 2008), we quantified fragmentation using a subset of metrics that have been
186 shown to characterize aspects of forest fragmentation important for forest-breeding birds (e.g.,
forest patch size variability and forest aggregation) (Table 2). The initial set of metrics used to
188 describe forest fragmentation included the percentage of landscape classified as forest (Forest),
area-weighted mean forest patch area (AREA_AM), forest patch area coefficient of variation
190 (AREA_CV), mean nearest neighbor distance (ENN), nearest neighbor distance coefficient of
variation (ENN_CV), area-weighted mean edge contrast (ECON), similarity index (SIMI), and
192 clumpiness index (CLUMPY) (McGarigal et al. 2002). Multicollinearity is a common problem
when using fragmentation metrics as predictors, and for each species we examined scatter plots
194 to identify the least correlated ($r < 0.5$) set of predictor variables. We included the proportion of
forest cover at a 100 km radius landscape (FOR) as a measure of regional habitat availability and
196 to account for processes that might be occurring over broader spatial scales and maximum
dispersal events (Tittler et al. 2009). We calculated all landscape metrics in FRAGSTATS
198 (McGarigal et al. 2002).

200 *Statistical analysis*

For each species, we used a generalized linear model with a Gaussian error distribution (GLM)
202 (Faraway 2006) and AIC_c-based model selection (Burnham and Anderson 2002) to identify the
most parsimonious model of the relationship between within-population genetic diversity and
204 forest fragmentation. Forest cover (at the 10 km landscape-scale), and regional forest availability
(at the 100 km landscape-scale) was included in every model to statistically examine the
206 variation in forest configuration while controlling for forest composition. All fragmentation

metrics were standardized for analysis. As the number of individuals sampled in a population can
208 potentially influence genetic measures (Landguth et al. 2010), we ran two-sided Grubb's tests for
each species to test for the presence of outliers in the number of sampled individuals (Grubbs
210 1950). A Grubb's test detects one outlier at a time, and the outliers are expunged from the
dataset and the test is iterated until no outliers are detected. Following the identification and
212 elimination of outliers, sample size (n) was included as a covariate in model selection. For all
model runs, we included a model that included only forest cover (at both landscape and regional
214 scales). We calculated the number of model parameters (K), AIC_c values, ΔAIC_c , and Akaike
weights (w_i). Models with a $\Delta AIC_c < 2$ were considered equivalent (Burnham and Anderson
216 2002). We calculated Nagelkerke R^2 for assessing the explained variation of the global model in
a likelihood framework (Nagelkerke 1991). We performed model selection using R (Version
218 2.10.1; R Development Core Team 2009) with the extension package MuMIn (Barton 2010).

220 *Spatial analysis*

We checked for spatial autocorrelation to quantify whether patterns in within-population genetic
222 diversity were associated with broader latitudinal gradients fragmentation (Spellman and Klicka
2006, 2007, Spellman et al. 2007) after accounting for the variation explained by forest cover
224 and fragmentation. We calculated spatial correlograms (Moran's I) on residuals of the model
with strongest support through model selection (Model 1 for each species) at different lag
226 distances (maximum lag distance = 200 km) and tested whether spatial autocorrelation was
significant through resampling with 10,000 iterations ($\alpha < 0.05$) (Dormann et al. 2007). We
228 performed all spatial analyses using the R extension package ncd (Dormann et al. 2007).

230 RESULTS

Genetic diversity within populations

232 We found within-population genetic diversity to be variable among populations (within-species)
both for haplotype diversity (*P. gambeli*: mean = 0.62 ± 0.26 ; *S. carolinensis*: mean = $0.77 \pm$
234 0.24 ; *S. pygmaea*: mean = 0.67 ± 0.23) and for pairwise nucleotide diversity (*P. gambeli*: mean =
 0.0019 ± 0.0015 ; *S. carolinensis*: mean = 0.0021 ± 0.0015 ; *S. pygmaea*: mean = $0.0021 \pm$
236 0.0010). As expected, haplotype diversity was correlated with nucleotide diversity for all three
species (*P. gambeli*: $r^2 = 0.41$, $P < 0.001$; *S. carolinensis*: $r^2 = 0.22$, $P = 0.04$; *S. pygmaea*: $r^2 =$
238 0.63 , $P < 0.001$).

240 *Patterns of forest fragmentation*

We found that the local 10 km landscapes surrounding the sampling locations consisted of two
242 primary land cover types, forested upland (characterized by natural or semi-natural wood
vegetation greater than 6 m tall) and shrubland habitats (characterized by natural or semi-natural
244 wood vegetation with aerial stems less than 6 m tall), and supported the analysis of these sites as
binary (forest vs. non-forest) landscapes. Shrubland (non-forest habitats) is a classification
246 representing natural vegetation (non-agricultural) as areas dominated by shrubs less than 5 m tall,
and is often also associated with grasses, sedges, herbs, and non-vascular vegetation.

248 For *P. gambeli* populations we found forest composition (10 km radius) ranging from
25% to 95% (mean = 62%) and regional forest composition (100 km) ranging from 0% to 70%
250 (mean = 21%). We found similar patterns for the landscapes surrounding *S. carolinensis*
populations, we found that forest composition ranged from 23% to 87% (mean = 61%) and
252 regional forest composition ranged from 1% to 50% (mean = 21%). *Sitta pygmaea* landscapes

were also characterized by a combination of forest and shrubland cover types with forest
254 composition ranging from 2% to 96% (mean = 50%) and regional forest composition from 0% to
65% (mean = 14%).

256

Genetic diversity and forest fragmentation

258 Normal-probability plots and histograms showed that normality for regression-model residuals
was met. For *P. gambeli*, sample size ranged from 4-19 individuals and the outlier test was
260 significant ($U = 0.40$, $P = 0.003$). As such, we iteratively removed populations with fewer than 5
and greater than 14 individuals sampled. Re-running the outlier test demonstrated that no more
262 outliers were detected. The uncorrelated set of metrics included AREA_CV and CLUMPY that
resulted in eight separate models. For haplotype diversity, we found that two out of four models
264 had support ($\Delta AIC_c < 2$; Table 3). The model with the strongest support ($w_i = 0.46$) included the
forest cover only model, while the second best model ($w_i = 0.18$) included only CLUMPY. For
266 *P. gambeli*, haplotype diversity was slightly lower in landscapes with higher forest cover at
landscape (Forest; $\beta = -0.13 \pm 0.22$) and regional scales (FOR; $\beta = -0.03 \pm 0.23$), but these
268 relationships were highly variable. When forest aggregation (CLUMPY) was included in the
model (Model 1; Table 3), we found that haplotype diversity for *P. gambeli* was higher in
270 landscapes with a higher degree of forest aggregation (CLUMPY; $\beta = 0.06 \pm 0.05$; Fig. 2A). The
Nagelkerke R^2 for the global model was 0.41. For nucleotide diversity, we found that one model
272 had the strongest support ($w_i = 0.60$). Similar to haplotype diversity, we found that nucleotide
diversity was higher in landscapes of higher forest aggregation (CLUMPY; $\beta = 0.79 \times 10^{-3} \pm$
274 0.29×10^{-3} ; Fig. 2B).

For *S. carolinensis*, sample size ranged from 5-11 individuals, no outliers were detected
276 ($U = 0.70$, $P = 1$), and the least correlated set of fragmentation metrics for *S. carolinensis* were
AREA_CV, SIMI, and CLUMPY. Combinations of these variables resulted in eight separate
278 models. For haplotype diversity, we found that two models had the highest support in model
selection ($w_i = 0.86$). The top model ($w_i = 0.57$) only included forest patch size variability as a
280 fragmentation metric (AREA_CV; Table 3). We found that *S. carolinensis* populations
inhabiting more fragmented landscapes with a higher variability of forest patch sizes (high
282 AREA_CV) had lower haplotype diversity ($\beta = -0.12 \pm 0.03$; Fig. 3). Haplotype diversity was
also slightly higher in more forested landscapes at regional scales ($\beta = 0.09 \pm 0.23$), although this
284 was highly variable. The second most supported model ($w_i = 0.29$), included sample size (n) and
showed a slightly positive relationship with haplotype diversity ($\beta = 0.03 \pm 0.01$). The
286 Nagelkerke R^2 for the global model was 0.78. When analyzing nucleotide diversity the model
including only forest cover had the most support ($w_i = 0.78$; Table 3) suggesting that none of the
288 fragmentation metrics provided sufficient explanatory power in describing the variation in
nucleotide diversity for *S. carolinensis*. In these models, nucleotide diversity was higher in more
290 forested landscapes at a local scale, but slightly lower in forested regions.

Sitta pygmaea populations ranged from 4-11 individuals sampled and the outlier test was
292 not significant ($U = 0.82$, $P = 1$). We included three uncorrelated fragmentation metrics
including AREA_CV, ENN_CV, and CLUMPY in model selection. Three out of six models had
294 support ($\Delta AIC_c < 2$), and the final model set included the model with forest cover only (Table
3). In contrast to *P. gambeli* and *S. carolinensis*, we found that the haplotype diversity of *S.*
296 *pygmaea* populations generally increased in more fragmented landscapes (Nagelkerke $R^2 =$
0.47). Haplotype diversity was lower in landscapes with higher forest aggregation (CLUMPY; β

298 = -0.09 ± 0.04 ; Fig. 4) and higher in landscapes with more forest patch size variability
(AREA_CV; $\beta = 0.10 \pm 0.04$). Nucleotide diversity showed the same general patterns including
300 the forest cover only with highest support, given the data ($w_i = 0.44$), and lower nucleotide
diversity in landscape characterized by higher forest aggregation (CLUMPY; $\beta = -0.31 \times 10^{-2} \pm$
302 0.02×10^{-2}).

304 *Spatial autocorrelation*

We did not find any evidence of systematic patterns of spatial autocorrelation in the model
306 residuals for any species (Fig. 5). After inspecting the spatial correlograms, we concluded that
once the models incorporated the effects of forest configuration and composition, there were no
308 broader spatial patterns in unexplained variation of within-population genetic diversity
associated with geographic proximity.

310

DISCUSSION

312 We used existing genetic data on three forest breeding bird species to examine the relationship
between within-population genetic diversity and forest fragmentation across multiple
314 independent landscapes. We predicted that patterns of within-population genetic diversity with
fragmentation would vary among species as a consequence of differences in their degree of
316 habitat specialization or dispersal abilities. However, these patterns did not vary with life history
traits in the directions that we predicted. Specifically, we predicted that within-population
318 genetic diversity would be more likely to decline in response to habitat fragmentation among
habitat specialists and among weak dispersers (Bailey et al. 2007) because they would be less
320 likely to disperse through the intervening matrix between appropriate habitat fragments (and,

therefore, would be more genetically isolated) than species with relatively broad habitat
322 associations and stronger dispersal abilities. For two of the three species (*P. gambeli* and *S.*
carolinensis), we found evidence that higher within-population genetic diversity was associated
324 with landscapes characterized by lower levels of forest fragmentation. We found forest
aggregation (CLUMPY) and the variability in forest patch area (AREA_CV) to be the two
326 strongest landscape predictors of genetic diversity in these two species. These metrics captured
the characteristics of habitat fragmentation, where large, compact clusters of forest with low
328 variability in size supported populations with higher levels of within-population genetic
diversity.

330 Our findings for *P. gambeli* and *S. carolinensis* are consistent with the predominant
conclusion of a survey of studies comparing within-population genetic diversity between
332 fragmented and control landscapes: in 26 of 45 comparisons (~58%), studies found reduced
genetic differentiation associated with fragmentation (Keyghobadi 2007). However, nearly half
334 of these studies ($n = 19$) found no effect or even an effect in the opposite direction (Keyghobadi
2007). Likewise, in this study, we found patterns of higher within-population genetic diversity of
336 *S. pygmaea* in fragmented landscapes. In our study, genetic diversity was higher in fragmented
landscapes for the species that was the most specialized (Kingery and Ghalambor 2001,
338 Spellman and Klicka 2006) and the most sedentary (Kingery and Ghalambor 2001), *S. pygmaea*.
In contrast, *P. gambeli* and *S. carolinensis* have arguably broader habitat associations and
340 stronger dispersal abilities than *S. pygmaea* (McCallum et al. 1999, Grubb and Pravosudov
2008), yet both showed a reduction of genetic diversity with fragmentation. Information about
342 dispersal distances for these species (and for songbirds in general) is limited, and is likely to be
greater than the distances reported in the literature (Tittler et al. 2009). Nevertheless, the patterns

344 that we observed did not match our predictions based on their putative dispersal distances
relative to one another, or with their relative specificity of habitat associations.

346 The elevated genetic diversity of *S. pygmaea* in fragmented landscapes is further
surprising because of their cooperatively breeding social system. Many breeding pairs have one
348 or more male helpers, usually offspring from previous broods (Kingery and Ghalambor 2001).
Some juveniles merge with neighbors from the previous winter's group (Guntert et al. 1988,
350 Sydeman et al. 1988). The short dispersal distances, natal philopatry, and habitat specificity of *S.*
pygmaea are typical of many cooperatively breeding species (Woxvold et al. 2006, Haas et al.
352 2010). Such characteristics likely contribute to geographic isolation and genetic structuring
among populations, and could increase the vulnerability of cooperative breeders to habitat loss
354 and fragmentation (Koenig et al. 1996, Walters et al. 2004, Haas et al. 2010). However,
cooperative breeders could be buffered from loss of genetic diversity in large fragments, from
356 which juveniles might not have to disperse to find a breeding vacancy (Walters et al. 2004).
More research is needed regarding how cooperative breeding could influence the relationship
358 between fragmentation and genetic diversity.

Studying the relationship between habitat and within-population genetic diversity using
360 contemporary landscape data can be problematic because variation quantified by molecular
markers such as mitochondrial DNA may reflect historic, rather than contemporary, patterns of
362 environmental variation (Frankham et al. 2010, Landguth et al. 2010, Sork and Waits 2010). As
an example, *S. pygmaea* exhibited a positive correlation between genetic diversity and increased
364 fragmentation and is the only species in the study with a single glacial refuge but a similarly
large contemporary distribution (Spellman and Klicka 2006). By contrast, *P. gambeli* and *S.*
366 *carolinensis* had lower diversity in fragmented landscapes and are phylogeographically

structured into two or more reciprocally monophyletic clades and expanded to occupy their
368 current distribution from two (or more) glacial refugia (Spellman and Klicka 2007, Spellman et
al. 2007). Given the difference of range expansion from a single versus multiple glacial refugia,
370 it is possible that the overall effective population size of *S. pygmaea* following expansion was
larger and the contemporary distribution was achieved later than in *P. gambeli* or *S. carolinensis*.
372 If this were true, then we would expect *S. pygmaea* to enter genetic drift equilibrium slower than
other species with smaller, more disjunct postglacial population size. Thus, *S. pygmaea* may not
374 have yet achieved this equilibrium, even though they have the lowest dispersal rates of the three
examined species.

376 In some studies, when both historic and contemporary land use data have been studied,
information about historic landscapes has proven to be a better predictor of current genetic
378 variation than contemporary habitat fragmentation (Keyghobadi et al. 2005a, 2005b, Holzhauer
et al. 2006). Mitochondrial markers generally reflect the genetic imprint of historical processes
380 occurring hundreds of years before present (Frankham et al. 2010), but the samples used in this
study were collected from natural areas less affected by modern day anthropogenic disturbance.
382 Hence, it is likely that the contemporary patterns and variability of fragmentation across the
study landscapes were sustained primarily by biophysical processes (e.g. soil type, topography,
384 moisture gradients) rather than caused by human activity (e.g., human development). That being
said, additional studies utilizing molecular markers (e.g., microsatellites) that may more readily
386 capture contemporary genetic patterns could provide further insight into the relationships
between patterns of landscape fragmentation and within-population genetic diversity we found.

388 Habitat fragmentation is a major driver of environmental change affecting avian
populations across multiple levels of diversity (Fahrig 2003). In recent years, landscape genetics

390 has focused on quantifying the effects of fragmentation and landscape connectivity on the
genetic structure among populations (Luque et al. 2012), but studies must continue to take a
392 landscape-scaled approach of quantifying how characteristics of habitat configuration are
correlated with within-population genetic variation among a wide range of landscapes. Using
394 genetic data from multiple populations across a broad geographic area, we found within-
population genetic diversity was generally higher in landscapes characterized by less fragmented
396 forest habitat for two out of three forest breeding birds. These findings support the prediction that
fragmented landscapes sustain genetically less diverse populations; however, we found the
398 opposite pattern for the species that would normally be considered most susceptible to
fragmentation effects due to its limited dispersal abilities and habitat specificity. For broader
400 generalization based on empirical data, studies on genetic diversity and multiple species of
varying life-history traits inhabiting multiple landscapes are needed.

402

ACKNOWLEDGMENTS

404 We thank the Carling/Benkman (University of Wyoming) and Dickinson/Dhondt (Cornell Lab of
Ornithology) lab groups for constructive suggestions on an earlier version of this manuscript.

406

LITERATURE CITED

- 408 Anderson, C. D., B. K. Epperson, M. J. Fortin, R. Holderegger, P. M. A. James, et al. 2010.
Considering spatial and temporal scale in landscape-genetic studies of gene flow. *Molecular*
410 *Ecology* 19:3565-3575.
- Avise, J. C. 2000. *Phylogeography: the history and formation of species*. Cambridge: Harvard
412 University Press. 447 p.
- Barton, K. 2010. MuMIn: Multi-model inference. R package version 0.13.9. [http://CRAN.R-](http://CRAN.R-project.org/package=MuMIn)
414 [project.org/package=MuMIn](http://CRAN.R-project.org/package=MuMIn).
- Bélisle, M., A. Desrochers, and M. J. Fortin. 2001. Influence of forest cover on the movements
416 of forest birds: a homing experiment. *Ecology* 82:1893-1904.
- Boulinier, T., J. D. Nichols, J. E. Hines, J. R. Sauer, C. H. Flather, et al. 1998. Higher temporal
418 variability of forest breeding bird communities in fragmented landscapes. *Proceedings of the*
National Academy of Sciences of the United States of America 95:7497-7501.
- 420 Boulinier, T., J. D. Nichols, J. E. Hines, J. R. Sauer, C. H. Flather, et al. 2001. Forest
fragmentation and bird community dynamics: Inference at regional scales. *Ecology* 82:1159-
422 1169.
- Bowie, R. C. K. 2011. The utility of contemporary and historical estimates of dispersal in
424 determining response to habitat fragmentation in a tropical forest-dependent bird community.
Molecular Ecology 20:1799-1802.
- 426 Brawn, J. D., and R. P. Balda. 1988. Population biology of cavity nesters in northern Arizona:
Do nest sites limit breeding densities? *Condor* 90:61-71.

- 428 Bruggeman, D. J., T. Wiegand, and N. Fernandez. 2010. The relative effects of habitat loss and
fragmentation on population genetic variation in the red-cockaded woodpecker (*Picoides*
430 *borealis*). *Molecular Ecology* 19:3679-3691.
- Burnham, K. P., and D. R. Anderson. 2002. Model selection and inference: a practical
432 information-theoretic approach. New York: Springer-Verlag.
- Callens, T., P. Galbusera, E. Matthysen, E. Y. Durand, M. Githiru, et al. 2011. Genetic signature
434 of population fragmentation varies with mobility in seven bird species of a fragmented Kenyan
cloud forest. *Molecular Ecology* 20:1829-1844.
- 436 Chalfoun, A. D., F. R. Thompson, and M. J. Ratnaswamy. 2002. Nest predators and
fragmentation: a review and meta-analysis. *Conservation Biology* 16:306-318.
- 438 Cushman, S. A., K. McGargal, and M. C. Neel. 2008. Parsimony in landscape metrics: Strength,
universality, and consistency. *Ecological Indicators* 8:691-703.
- 440 Debinski, D. M., and R. D. Holt. 2000. A survey and overview of habitat fragmentation
experiments. *Conservation Biology* 14:342-355.
- 442 Donovan, T. M., and C. H. Flather. 2002. Relationships among north American songbird trends,
habitat fragmentation, and landscape occupancy. *Ecological Applications* 12:364-374.
- 444 Dormann, C. F., J. M. McPherson, M. B. Araujo, R. Bivand, J. Bolliger, et al. 2007. Methods to
account for spatial autocorrelation in the analysis of species distributional data: A review.
446 *Ecography* 30:609-628.
- Excoffier L., G. Laval, and S. Schneider. 2005. Arlequin (version 3.0): An integrated software
448 package for population genetics data analysis. *Evolutionary Bioinformatics* 1:47-50.
- Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology*
450 *Evolution and Systematics* 34:487-515.

- Faraway, J. J. 2006. Extending the linear model with R: Generalized linear, mixed effects and
452 nonparametric regression models. Boca Raton: Chapman & Hall/CRC. 301 p.
- Frankham, R., J. D. Ballou, and D. A. Briscoe. 2010. Introduction to conservation genetics.
454 Cambridge: Cambridge University Press. 618 p.
- Grubb, T. C., and V. V. Pravosudov. 2008. White-breasted Nuthatch (*Sitta carolinensis*). In A.
456 Poole [ed.], The Birds of North America Online: <http://bnabirds.cornelledu/bna/species/054>
Accessed 13 February 2013.
- 458 Grubbs, F. E. 1950. Sample criteria for testing outlying observations. *Annals of Mathematical
Statistics* 21:27-58.
- 460 Güntert, M., D. B. Hay, and R. P. Balda. 1988. Communal roosting in pygmy nuthatches: A
winter survival strategy. *Proceedings of the 19th International Ornithological Congress* 19:1964-
462 1972.
- Haas, S. E., J. A. Cox, J. V. Smith, and R. T. Kimball. 2010. Fine-scale spatial genetic structure
464 in the cooperatively breeding Brown-headed Nuthatch (*Sitta pusilla*). *Southeastern Naturalist*
9:743-756.
- 466 Hartl, D. L., and A. G. Clark. 2007. Principles of population genetics. Sunderland: Sinauer
Associates. 652 p.
- 468 Holzhauer, S. I. J., K. Ekschmitt, A. C. Sander, J. Dauber, and V. Wolters. 2006. Effect of
historic landscape change on the genetic structure of the bush-cricket *Metrioptera roeseli*.
470 *Landscape Ecology* 21:891-899.
- Homer, C., J. Dewitz, J. Fry, M. Coan, N. Hossain, et al. 2007. Completion of the 2001 National
472 Land Cover Database for the conterminous United States. *Photogrammetric Engineering and
Remote Sensing* 73:337-341.

- 474 Jaarola, M., H. Tegelstrom, and K. Fredga. 1999. Colonization history in Fennoscandian rodents.
Biological Journal of the Linnean Society 68:113-127.
- 476 Jenkins, D. G., M. Carey, J. Czerniewska, J. Fletcher, T. Hether, et al. (2010). A meta-analysis of
isolation by distance: relic or reference standard for landscape genetics? *Ecography* 33:315-320.
- 478 Johansson, M., C. R. Primmer, J. Sahlsten, and J. Merila. 2005. The influence of landscape
structure on occurrence, abundance and genetic diversity of the common frog, *Rana temporaria*.
480 *Global Change Biology* 11:1664-1679.
- Keyghobadi, N. 2007. The genetic implications of habitat fragmentation. *Canadian Journal of*
482 *Zoology* 85:1049-1064.
- Keyghobadi, N., J. Roland, S. F. Matter, and C. Strobeck. 2005. Among- and within-patch
484 components of genetic diversity respond at different rates to habitat fragmentation: An empirical
demonstration. *Proceedings of the Royal Society B-Biological Sciences* 272:553-560.
- 486 Keyghobadi, N., J. Roland, and C. Strobeck. 2005. Genetic differentiation and gene flow among
populations of the alpine butterfly, *Parnassius smintheus*, vary with landscape connectivity.
488 *Molecular Ecology* 14:1897-1909.
- Kingery, H. E., and C. K. Ghalambor. 2001. Pygmy Nuthatch (*Sitta pygmaea*). In A. Poole [ed.],
490 *The Birds of North America Online*: <http://bnabirds.cornell.edu/bna/species/567> Accessed 13
February 2013.
- 492 Koenig, W. D., D. VanVuren, and P. N. Hooge. 1996. Detectability, philopatry, and the
distribution of dispersal distances in vertebrates. *Trends in Ecology and Evolution* 11:514-517.
- 494 Landguth, E. L., S. A. Cushman, M. K. Schwartz, K. S. McKelvey, M. Murphy, et al. (2010).
Quantifying the lag time to detect barriers in landscape genetics. *Molecular Ecology* 19:4179-
496 4191.

- Luque, S., S. Saura, and M. J. Fortin. 2012. Landscape connectivity analysis for conservation:
498 Insights from combining new methods with ecological and genetic data. *Landscape Ecology*
27:153-157.
- 500 Matthysen, E., and K. H. Schmidt. 1987. Natal dispersal in the Nuthatch. *Ornis Scandinavica*
18:313-316.
- 502 McCallum, D. A., R. Grundel, and D. L. Dahlsten. 1999. Mountain Chickadee (*Poecile gambeli*).
In A. Poole [ed.], *The Birds of North America Online*: <http://bnabirds.cornelledu/bna/species/453>
504 Accessed 13 February 2013.
- McGarigal, K., and S. A. Cushman. 2002. Comparative evaluation of experimental approaches to
506 the study of habitat fragmentation effects. *Ecological Applications* 12:335-345.
- McGarigal, K., S. A. Cushman, M. C. Neel, and E. Ene. 2002. FRAGSTATS: Spatial Pattern
508 Analysis Program for Categorical Maps. Computer software program produced by the authors at
the University of Massachusetts, Amherst. Available at the following web site:
510 www.umass.edu/landeco/research/fragstats/fragstats.html.
- Mech, S. G., and J. G. Hallett. 2001. Evaluating the effectiveness of corridors: a genetic
512 approach. *Conservation Biology* 15:467-474.
- Møller, A. P., L. Z. Garamszegi, and C. N. Spottiswoode. 2008. Genetic similarity, breeding
514 distribution range and sexual selection. *Journal of Evolutionary Biology* 21:213-225.
- Nagelkerke, N. J. D. 1991. A note on a general definition of the coefficient of determination.
516 *Biometrika* 78:691-692.
- Norris, R. A. 1958. Comparative biosystematics and life history of the nuthatches *Sitta pygmaea*
518 and *Sitta pusilla*. *University of California Publications in Zoology* 56:119-300.

- R Development Core Team. 2009. R: A language and environment for statistical computing.
520 Vienna: R Foundation for Statistical Computing.
- Sork, V. L., and L. Waits. 2010. Contributions of landscape genetics – approaches, insights, and
522 future potential. *Molecular Ecology* 19:3489-3495.
- Spellman, G. M., and J. Klicka. 2007. Phylogeography of the white-breasted nuthatch (*Sitta*
524 *carolinensis*): diversification in North American pine and oak woodlands. *Molecular Ecology*
16:1729-1740.
- 526 Spellman, G. M., B. Riddle, and J. Klicka. 2007. Phylogeography of the mountain chickadee
(*Poecile gambeli*): diversification, introgression, and expansion in response to Quaternary
528 climate change. *Molecular Ecology* 16:1055-1068.
- Spellman, G. M., and J. Klicka. 2006. Testing hypotheses of Pleistocene population history using
530 coalescent simulations: phylogeography of the Pygmy Nuthatch (*Sitta pygmaea*). *Proceedings of*
the Royal Society B-Biological Sciences 273:3057-3063.
- 532 Storfer, A., M. A. Murphy, S. F. Spear, R. Holderegger, and L. P. Waits. 2010. Landscape
genetics: Where are we now? *Molecular Ecology* 19:3496-3514.
- 534 Sydeman, W. J., M. Güntert, and R. P. Balda. 1988. Annual reproductive yield in the cooperative
Pygmy Nuthatch (*Sitta pygmaea*). *Auk* 105:70-77.
- 536 Templeton, A. R., K. Shaw, E. Routman, and S. K. Davis. 1990. The genetic consequences of
habitat fragmentation. *Annals of the Missouri Botanical Garden* 77:13-27.
- 538 Thomassen, H. A., Z. A. Cheviron, A. H. Freedman, R. J. Harrigan, R. K. Wayne, et al. 2010.
Spatial modelling and landscape-level approaches for visualizing intra-specific variation.
540 *Molecular Ecology* 19:3532-3548.

- Tittler, R., M. A. Villard, and L. Fahrig. 2009. How far do songbirds disperse? *Ecography* 32:1051-1061.
- Turner, M. G., R. H. Gardner, and R. V. O'Neill. 2001. *Landscape ecology in theory and practice: Pattern and process*. New York: Springer. 401 p.
- Van Houtan, K. S., S. L. Pimm, J. M. Halley, R. O. Bierregaard, and T. E. Lovejoy. 2007. Dispersal of Amazonian birds in continuous and fragmented forest. *Ecology Letters* 10:219-229.
- Vandergast, A. G., A. J. Bohonak, D. B. Weissman, and R. N. Fisher. 2007. Understanding the genetic effects of recent habitat fragmentation in the context of evolutionary history: phylogeography and landscape genetics of a southern California endemic Jerusalem cricket (Orthoptera: Stenopelmatidae: *Stenopelmatus*). *Molecular Ecology* 16:977-992.
- Villard, M. A., M. K. Trzcinski, and G. Merriam. 1999. Fragmentation effects on forest birds: relative influence of woodland cover and configuration on landscape occupancy. *Conservation Biology* 13:774-783.
- Walters, J. R., C. B. Cooper, S. J. Daniels, G. Pasinelli, and K. Schiegg. 2004. Conservation biology. *In* W. D. Koenig and J. J. Dickinson [eds.], *Ecology and Evolution of Cooperative Breeding in Birds* Cambridge: Cambridge University Press. pp. 197-209.
- Wilcove, D. S. 1985. Nest predation in forest tracts and the decline of migratory songbirds. *Ecology* 66:1211-1214.
- Winkel, W. 1989. Distribution factors and age structure of the nuthatch (*Sitta eueopae caesia*). *Vogelwarte* 35:37-48.
- Woxvold, I. A., G. J. Adcock, and R. A. Mulder. 2006. Fine-scale genetic structure and dispersal in cooperatively breeding apostlebirds. *Molecular Ecology* 15:3139-3146.

Young, A., T. Boyle, and T. Brown. 1996. The population genetic consequences of habitat
564 fragmentation for plants. *Trends in Ecology and Evolution* 11:413-418.

566 Figure Legends

FIGURE 1. Geographic locations of sampling points for *Poecile gambeli* (A.), *Sitta carolinensis*
568 (B.), and *Sitta pygmaea* (C.) in western USA. Labels represent population identifiers (see Table
1).

570

FIGURE 2. For *P. gambeli*, haplotype diversity (A.) and nucleotide diversity (B.) was higher in
572 landscapes characterized by a higher degree of forest aggregation (CLUMPY).

574 FIGURE 3. For *S. carolinensis* populations, haplotype diversity was lower in more fragmented
landscapes characterized by higher variation in forest patch size (AREA_CV).

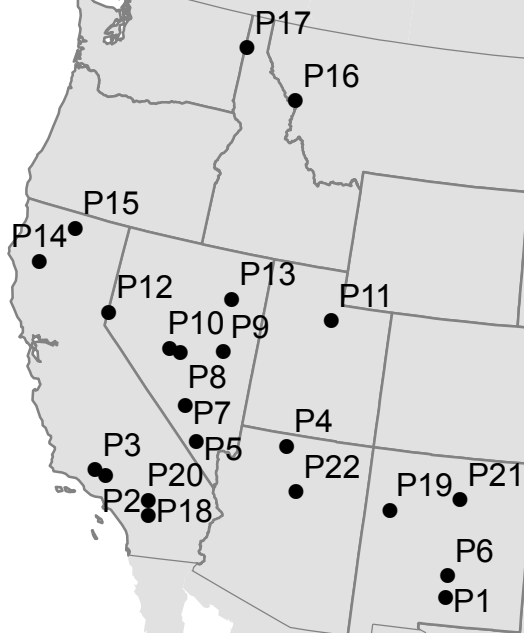
576

FIGURE 4. For *S. pygmaea*, haplotype diversity was lower in landscapes characterized by a
578 higher degree of forest aggregation (CLUMPY).

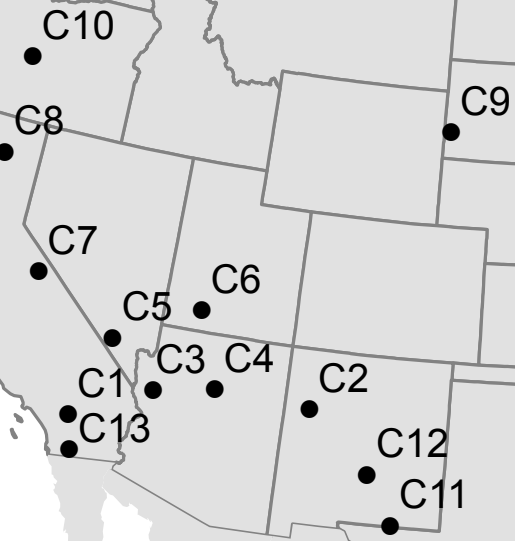
580 FIGURE 5. Spatial correlograms of GLM model residuals for *P. gambeli* (circles), *S.*
carolinensis (squares) and *S. pygmaea* (triangles) for haplotype (A.) and nucleotide (B.)
582 diversity. Filled symbols indicate significant spatial autocorrelation ($P < 0.05$) based on 10,000
permutations within a given distance class.

584

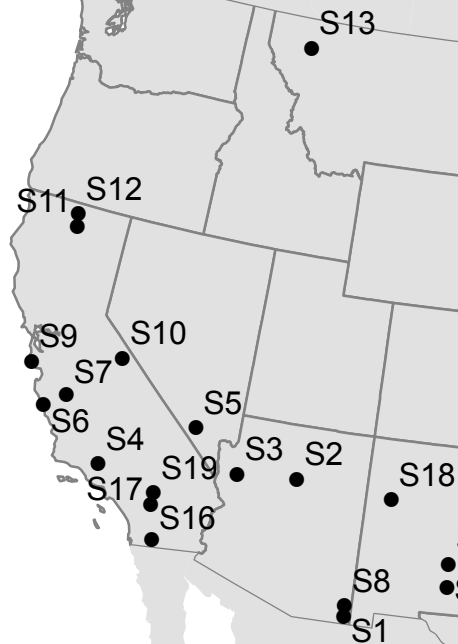
A.

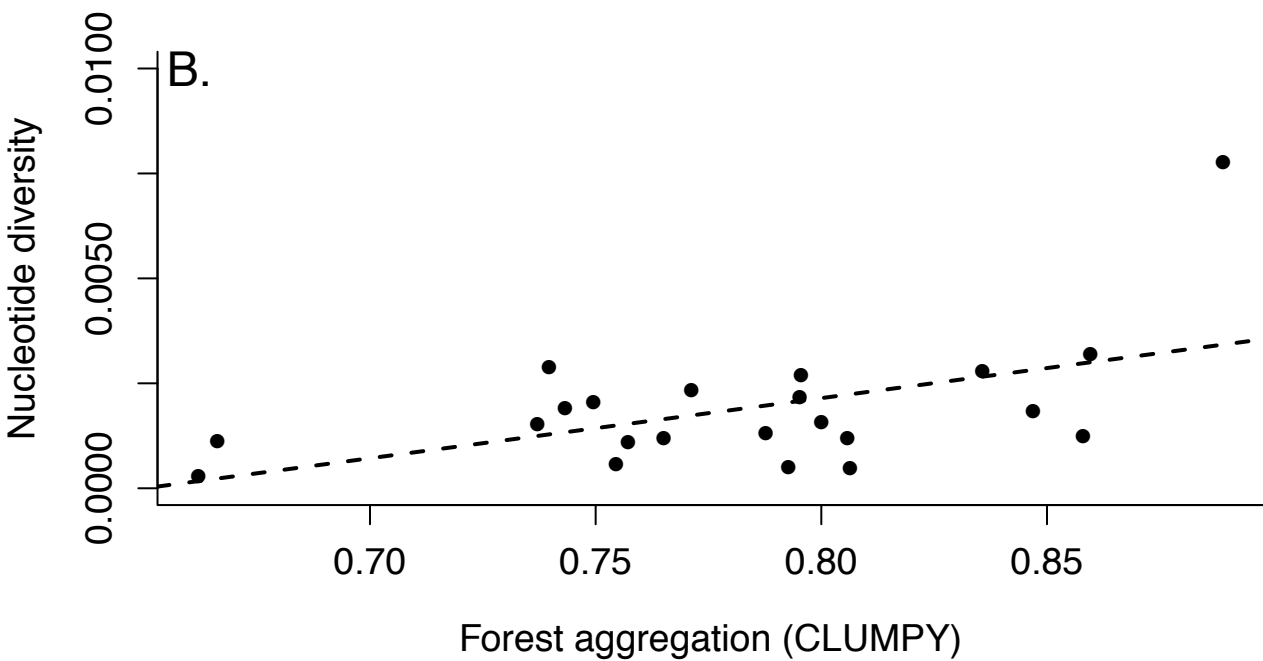
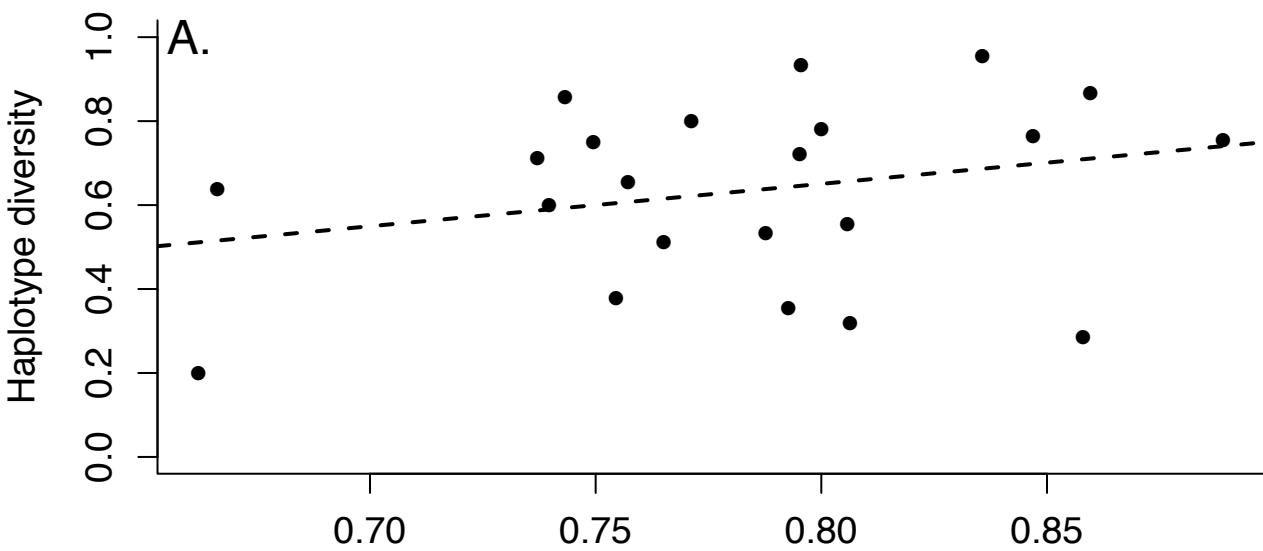


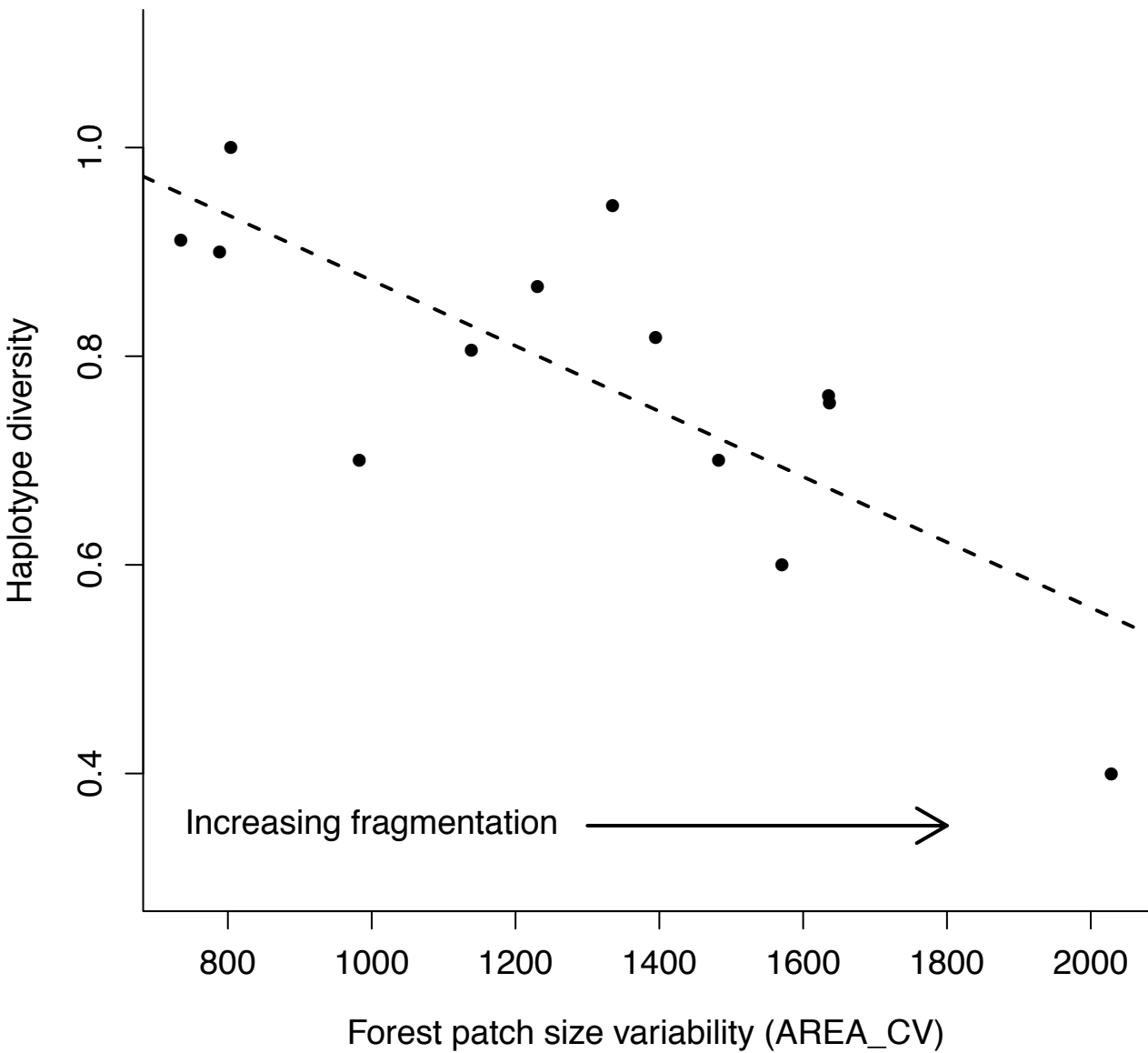
B.

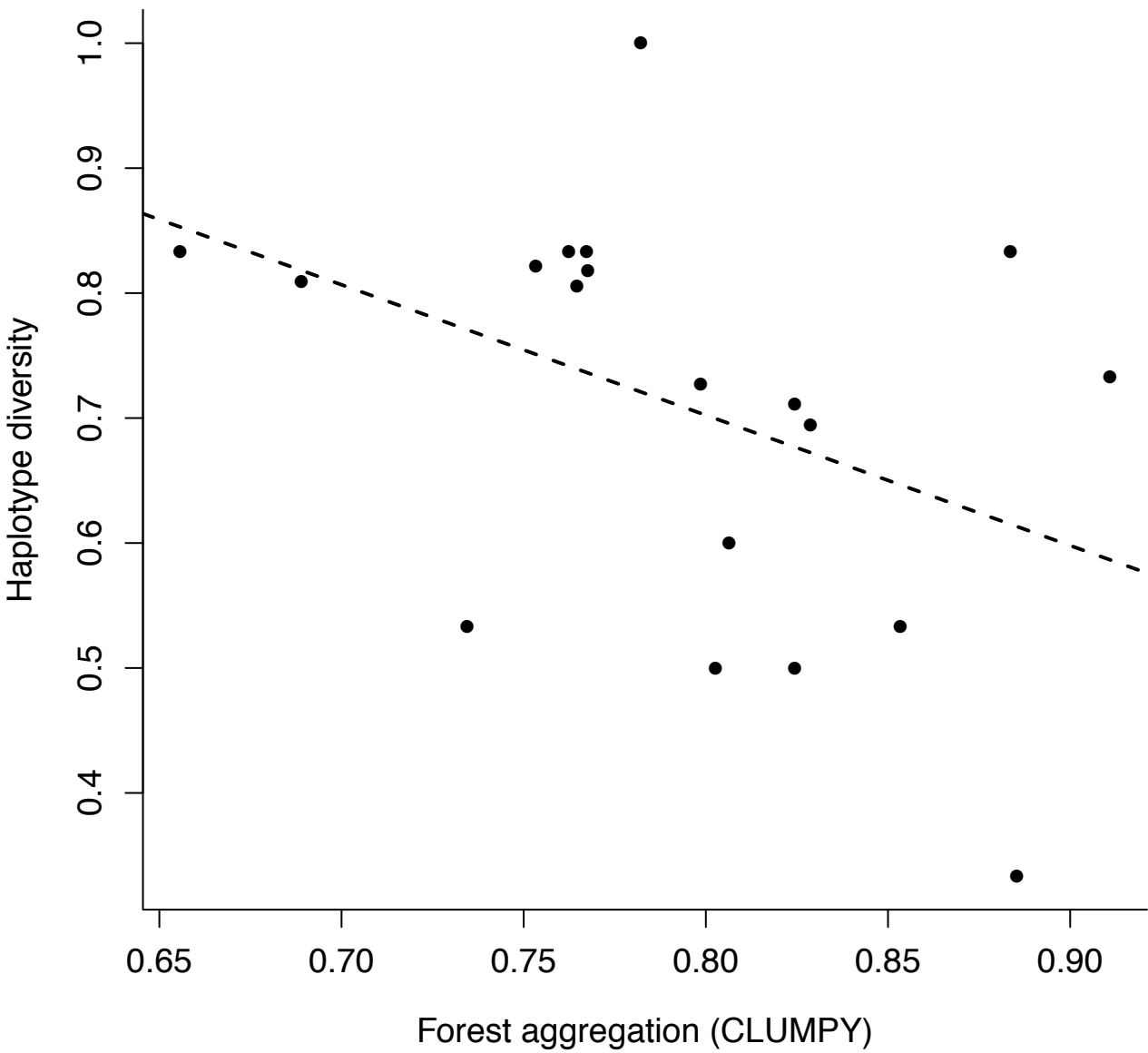


C.









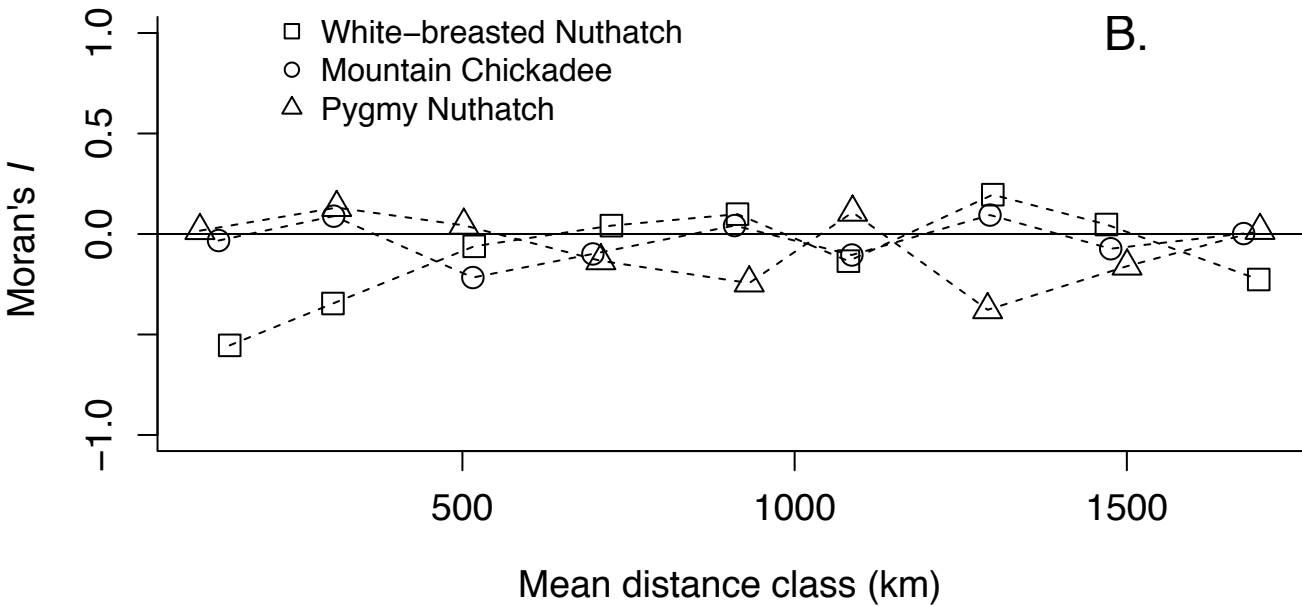
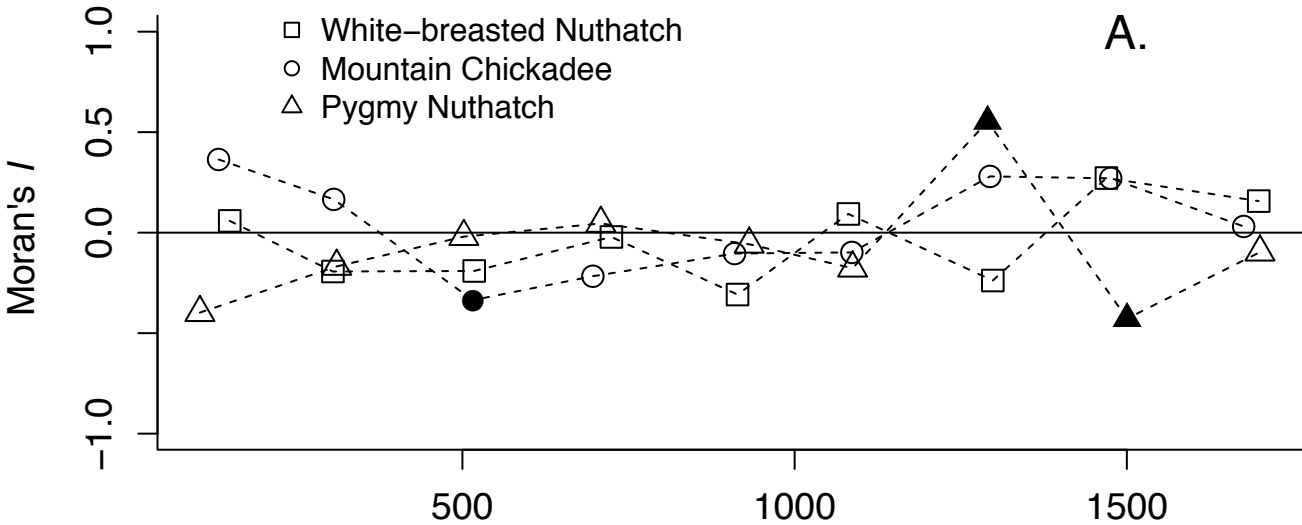


TABLE 1. Population identifier, number of individuals sampled (n), haplotype number (number of unique haplotypes in each population), haplotype diversity and standard deviation, and pairwise nucleotide diversity and standard deviation (SD), derived from complete sequences of the mitochondrial gene NADH dehydrogenase subunit 2 (ND2), for each population from the three species used in this study.

Pop	n	Number of Haplotypes	Haplotype Diversity	Haplotype Diversity (SD)	Nucleotide Diversity	Nucleotide Diversity (SD)
<i>P. gambeli</i>						
P1	8	4	0.75	0.1391	0.002058	0.001591
P2	10	3	0.60	0.1305	0.002882	0.002002
P3	10	4	0.53	0.1801	0.001313	0.001119
P4	9	3	0.64	0.1258	0.001121	0.001017
P5	10	5	0.80	0.1001	0.002337	0.001702
P6	7	2	0.29	0.1964	0.001235	0.001126
P7	10	8	0.96	0.0594	0.002786	0.001950
P8	10	7	0.87	0.1072	0.003202	0.002177
P9	10	5	0.76	0.1295	0.007781	0.004633
P10	10	7	0.93	0.0620	0.002690	0.001897
P11	10	2	0.36	0.1591	0.000512	0.000612
P12	10	3	0.51	0.1643	0.001185	0.001043
P13	10	3	0.71	0.0860	0.001537	0.001250
P14	10	2	0.20	0.1541	0.000288	0.000439
P15	10	3	0.38	0.1813	0.000576	0.000657
P16	7	4	0.86	0.1023	0.001921	0.001544
P17	11	5	0.78	0.0926	0.001572	0.001257
P18	11	5	0.76	0.1066	0.001834	0.001405
P19	12	3	0.31	0.1637	0.00048	0.000578
P20	13	4	0.65	0.1060	0.001108	0.000972
P21	9	3	0.56	0.1653	0.001201	0.001065
P22	9	5	0.72	0.1592	0.002161	0.001624
<i>S. carolinensis</i>						
C1	5	2	0.60	0.1753	0.001153	0.001020
C2	5	3	0.70	0.2184	0.001537	0.001267
C3	5	2	0.40	0.2373	0.001153	0.001020
C4	11	7	0.82	0.1191	0.001537	0.001111
C5	9	7	0.94	0.0702	0.004696	0.002865
C6	5	5	1.00	0.1265	0.001921	0.001509
C7	6	4	0.87	0.1291	0.001153	0.000980
C8	7	3	0.76	0.1148	0.001006	0.000864

C9	5	3	0.70	0.2184	0.005764	0.003866
C10	5	4	0.90	0.1610	0.001921	0.001509
C11	10	5	0.76	0.1295	0.000982	0.000808
C12	9	5	0.81	0.1196	0.001067	0.000868
C13	10	7	0.91	0.0773	0.002092	0.001428
				<i>S. pygmaea</i>		
S1	4	3	0.83	0.2224	0.001921	0.001612
S2	11	6	0.73	0.1444	0.001816	0.001264
S3	4	2	0.50	0.2652	0.000480	0.000595
S4	4	4	1.00	0.1768	0.003842	0.002888
S5	9	4	0.69	0.1470	0.001548	0.001143
S6	4	3	0.83	0.2224	0.004003	0.002993
S7	6	2	0.53	0.1721	0.001537	0.001215
S8	8	4	0.82	0.1007	0.001818	0.001316
S9	6	3	0.73	0.1552	0.003202	0.002203
S10	10	4	0.71	0.1175	0.002263	0.001521
S11	4	3	0.83	0.2224	0.003202	0.002465
S12	4	2	0.50	0.2652	0.001921	0.001612
S13	4	3	0.83	0.2224	0.002081	0.001719
S14	7	4	0.81	0.1298	0.002379	0.001668
S15	6	2	0.33	0.2152	0.001281	0.001059
S16	11	7	0.89	0.1191	0.002987	0.001892
S17	10	4	0.53	0.1801	0.001153	0.000907
S18	11	4	0.60	0.1539	0.001991	0.001359
S19	9	5	0.81	0.1196	0.002295	0.001559

590

592

594

596

598

TABLE 2. Metrics used to quantify forest fragmentation within 10 km radius-landscapes. We include interpretations for only those metrics retained in the top models identified through model selection.

Metric	Description	Ecological Interpretation ^a
Forest	Percentage of landscape classified as forest in a 10-km landscape	Degree that landscape is comprised of forest
FOR	Percentage of landscape classified as forest in a 100-km landscape	Degree that surrounding region is comprised of forest
AREA_CV	Patch area coefficient of variation	Forest patch size variability, where increasing variability indicates greater fragmentation
CLUMPY	Clumpy index	Aggregation of forest patches, where positive index values indicate large, compact clusters of forest

^aInterpretation based on Cushman et al. (2008)

600

602

TABLE 3. Model selection results testing the relationship between fragmentation and genetic diversity for *P. gambeli*, *S. carolinensis* and *S. pygmaea*. We report Akaike's second-order information criterion (AIC_c), Akaike weights (w_i) and associated parameter estimates for the fragmentation metrics included in each model. Only models with strong support ($\Delta AIC_c < 2$) are presented. Forest composition at the 10 km (Forest) and 100 km (FOR) landscape scales.

	AIC_c	ΔAIC_c	w_i	n	Forest	FOR	AREA_CV	CLUMPY
<i>P. gambeli</i>								
<i>Haplotype diversity</i>								
Forest cover only	4.1	0.00	0.46		-0.13	-0.03		
Model 1	6.0	1.92	0.18		-0.12	0.01		0.06
<i>Nucleotide Diversity</i>								
Model 1	-217.1	0.00	0.60		0.30×10^{-3}	-0.75×10^{-3}		-0.75×10^{-3}
<i>S. carolinensis</i>								
<i>Haplotype diversity</i>								
Model 1	-4.9	0.00	0.57		-0.02	0.09	-0.12	
Model 2	-3.6	1.35	0.29	0.03	0.00	0.25	-0.11	
<i>Nucleotide diversity</i>								

Forest cover only	-121.5	0.00	0.78	0.23×10^{-2}	-0.26×10^{-2}		
<i>S. pygmaea</i>							
<i>Haplotype diversity</i>							
Model 1	3.8	0.00	0.25	-0.16	0.19		-0.09
Forest cover only	3.9	0.13	0.23	-0.07	0.13		
Model 2	4.1	0.33	0.21	-0.19	0.30	0.09	-0.13
<i>Nucleotide diversity</i>							
Forest cover only	-111.7	0.00	0.44	0.36×10^{-2}	0.74×10^{-2}		
Model 1	-109.8	1.88	0.17	0.07×10^{-2}	0.96×10^{-2}		-0.31×10^{-2}
