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Avian conservation value of pine plantation forests in northern Vietnam

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Summary

Decline of native forest cover is a worldwide concern. Recently, overall forest cover in Vietnam has increased, but most of the increase has been attributed to plantations of non-native trees. The conservation value of these plantations for birds is unknown. We compared avian species richness in pine plantations to that in second-growth and mature native forests in Tam Dao National Park, Vietnam. Bird species were classified into two categories: forest specialists or forest generalists. To account for strong heterogeneity in detection probabilities, the number of species in each category was estimated using the Pledger-Huggins estimator. We estimated total species richness and number of forest specialist species to be highest in mature forest (191; 95% CI = 96, 287, and 88; 95% CI = 47, 129 respectively), lower in second-growth forest (158; 95% CI = 87, 245 and 58; 95% CI = 18, 98 respectively), and lowest in pine plantation (106; 95% CI = 52, 158 and 49; 95% CI = 2, 97 respectively). The estimated number of forest generalist species was similar between mature forest and second-growth forest (103; 95% CI = 17, 189 and 100; 95% CI = 42, 158, respectively) and least in pine plantation (57; 95% CI = 31, 82). The maintenance of native forest types should receive priority for conservation in Vietnam and pine plantations should be managed to provide additional structure in the hope of increasing species richness.

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J. Barlow , L. A. M. Mestre , T. A. Gardner and A. P Carlos . (2007) The value of primary, secondary and plantation forests for Amazonian birds. *Conserv. Biol*. 136: 212–231.

H. Beukema , F. Danielsen , G. Vincent , S. Hardiwinoto and J. van Andel (2007) Plant and bird diversity in rubber agroforests in the lowlands of Sumatra, Indonesia. *Agroforest.Syst*. 70: 217–242.

T. Boulinier , J. D. Nichols , J. R. Sauer , J. E. Hines , and K. H Pollock . (1998) Estimating species richness: The importance of heterogeneity in species detectability. *Ecology* 79: 1018–1028.

L. A. Brand , G. C. White , and B. R Noon . (2008) Factors influencing species richness and community composition of breeding birds in a desert riparian corridor. *Condor* 110: 199–210.

K. P. Burnham and W. S Overton . (1978) Estimation of the size of a closed population when capture probabilities vary among animals. *Biometrika* 65: 625–633.

A. Chao (1987) Estimating the population size for capture-recapture data with unequal catchability. *Biometrics* 43: 783–791.

K. L. Cockle , M. L. Leonard and A. A Bodrati . (2005) Presence and abundance of birds in an Atlantic forest reserve and adjacent plantation of shade-grown yerba mate in Paraguay. *Biodivers. Conserv*. 14: 3265–3288.

R. Greenberg , P. Bichier , A. C. Angon , and R. Reitsma (1997) Bird populations in shade and sun coffee plantations in central Guatemala. *Conserv. Biol*. 11: 448–459.

R. M Huggins . (1991) Some practical aspects of a conditional likelihood approach to capture experiments. *Biometrics* 47: 725–732.

J. B. Hughes , G. C. Daily and P. R Ehrlich . (2002) Conservation of tropical forest birds in countryside habitats. *Ecol. Lett*. 5: 121–129.

T. N. Johnson , R. D. Applegate , D. E. Hoover , P. S. Gipson and B. K Sandercock . (2009) Evaluating avian community dynamics in restored riparian habitats with mark-recapture models. *Wilson J. Ornith*. 121: 22–40.

H. K. Kwok and R. T Corlett . (2000) The bird communities of a natural secondary forest and a Lophostemon confertus plantation in Hong Kong, South China. *For. Ecol. Manage*. 130: 227–234.

R. MacArthur and J. W MacArthur . (1961) On bird species-diversity. *Ecology* 42: 594–598.

J. D. Nichols , T. Boulinier , J. E. Hines , K. H. Pollock and J. R Sauer . (1998) Inference methods for spatial variation in species richness and community composition when not all species are detected. *Conserv. Biol*. 12: 1390–1398.

S. Pledger (2000) Unified maximum likelihood estimates for closed capture-recapture models using mixtures. *Biometrics* 56: 434–442.

T. R. S. Raman and R. Sukumar (2002) Responses of tropical rainforest birds to abandoned plantations, edges, and logged forest in the Western Ghats, India. *Anim. Conserv*. 5: 201–216.

R. Reitsma , J. D. Parrish and W. McLarney (2001) The role of cacao plantations in maintaining forest avian diversity in southeastern Costa Rica. *Agroforest. Syst*. 53: 185–193.

J. A Rotenberg . (2007) Ecological role of a tree (Gmelina arborea) plantation in Guatemala: an assessment of an alternative land use for tropical avian conservation. *Auk* 124: 316–330.

J. A. Royle and J. D Nichols . (2003) Estimating abundance from repeated presence-absence data or point counts. *Ecology* 84: 777–790.

F. E. Schwab , N. P. P. Simon and A. R. E Sinclair . (2006) Bird-vegetation relationships in southeastern British Columbia. *J. Wildl. Manage*. 70: 189–197.

N. S. Sodhi and B. W Brook . (2006) Southeast Asian birds in peril. *Auk* 123: 275–277.

N. S. Sodhi , M. R. C. Posa , T. M. Lee and I. G Warkentin . (2008) Effects of disturbance or loss of tropical rainforest on birds. *Auk* 125: 511–519.

B. A. Walther and J. L Moore . (2005) The concepts of bias, precision and accuracy, and their use in testing the performance of species richness estimators, with a literature review of estimator performance. *Ecography* 28: 815–829.

B. A. Walther and S. Morand (1998) Comparative performance of species richness estimation methods. *Parasitology* 116: 395–405.

G. C. White and K. P Burnham . (1999) Program MARK: survival estimation from populations of marked animals. *Bird Study* 46: 120–139.

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Summary

Decline of native forest cover is a worldwide concern. Recently, overall forest cover in Vietnam has increased, but most of the increase has been attributed to plantations of non-native trees. The conservation value of these plantations for birds is unknown. We compared avian species richness in pine plantations to that in second-growth and mature native forests in Tam Dao National Park, Vietnam. Bird species were classified into two categories: forest specialists or forest generalists. To account for strong heterogeneity in detection probabilities, the number of species in each category was estimated using the Pledger-Huggins estimator. We estimated total species richness and number of forest specialist species to be highest in mature forest (191; 95% CI = 96, 287, and 88; 95% CI = 47, 129 respectively), lower in second-growth forest (158; 95% CI = 87, 245 and 58; 95% CI = 18, 98 respectively), and lowest in pine plantation (106; 95% CI = 52, 158 and 49; 95% CI = 2, 97 respectively). The estimated number of forest generalist species was similar between mature forest and second-growth forest (103; 95% CI = 17, 189 and 100; 95% CI = 42, 158, respectively) and least in pine plantation (57; 95% CI = 31, 82). The maintenance of native forest types should receive priority for conservation in Vietnam and pine plantations should be managed to provide additional structure in the hope of increasing species richness.

Introduction

The extreme reduction of native forest cover worldwide is a current cause of concern (Collar *et al.* 1994, Sodhi *et al.* 2008). Declines in native forest cover have been observed in South-East Asia. In Vietnam, these declines have been largely attributed to the expansion of agriculture, logging, and the transformation of natural forest into plantations. These declines have led to the local extinction of many bird populations (Sodhi and Brook 2006). In the forests of northern Vietnam there are probably 46 species listed as threatened on the IUCN Red List (Lepage 2010). Recently, overall forest cover in Vietnam has increased, but most of the increase has been attributed to plantations of non-native trees, and natural forests are still being reduced and replaced by other land uses (Nhat 2001). Pines *Pinus* spp. are widely used as plantation species in Vietnam. The purpose of these pine plantations is to increase overall vegetation cover for wood and for turpentine production.

The conservation value of these plantations for birds is unknown, but few native forest bird species are assumed to persist in other land uses (Hughes *et al.* 2002). Bird species diversity has been found to be lower in some tree plantations relative to native forest types (Cockle *et al.* 2005, Barlow *et al.* 2007, Rotenberg 2007, Holbech 2009). For tree plantations, this may be due to low structural complexity because of uniform age and physiognomy of plantation trees and a poorly developed understorey, which may lead to lower abundance of food items and fewer opportunities for concealment for native birds (Kwok and Corlett 2000). In general, bird species diversity is reported to be positively correlated with vegetation structure and composition (MacArthur and MacArthur 1961, Wiens 1992). Recent evidence suggests that some land uses, such as tree

plantations, have potential for conservation of native forest bird species, especially when management practices diversify forest vegetation and composition (Reitsma *et al.* 2001, Beukema *et al.* 2007).

Managing industrial forests in Vietnam for the dual purposes of wood products and conservation is of wide national interest. Measures outlined in the recently proposed plan "Five Million Hectares of Forest" (Vietnam Government 1998) would increase the forest cover nationwide in Vietnam from 33% to 43% by planting non-native trees. The overall benefits of this plan will be enhanced if conservation values can be balanced with economic concerns. Our objective was to examine the avian conservation potential of pine plantations compared to other native vegetation types, including second-growth forests and mature forests. Specifically, we tested several predictions relating to patterns in the number of bird species (i.e. species richness) with forest type. We examined predictions concerning forest specialist and forest generalist species guilds. Forest specialists are bird species that are assumed to be intolerant of forest disturbance. Forest generalists are bird species that inhabit all forest types and are assumed to be more tolerant of forest disturbance. Mature native forests have very complex vegetation composition and structure, and are hypothesised to support a larger number of bird species, especially forest specialists, than plantation forests. Therefore, we predicted that species richness of forest specialists will be higher in mature forest than the other two vegetation types and that forest generalist species will be equally rich in mature forests, second-growth forests, and pine plantations. Additionally, we predicted that total species richness will be highest in mature forests and lower in second-growth forests and pine plantations. One problem with estimating species richness is that strong heterogeneity in detection of species can bias richness estimates (Boulinier *et al.* 1998). To address this problem we used model developments by Huggins (1991) and Pledger (2000).

Methods

Study area

The study area was in Tam Dao National Park (TDNP), Vietnam (21°21'–21°42'N, 105°23'–105°44'E). The main part of TDNP is in the Tam Dao mountain range with the highest peak reaching 1,590 m. The park is 10–15 km wide, 80 km long, and located in three provinces: Thai Nguyen, Tuyen Quang, Vinh Phuc. The park centre is 70 km north-west of the capital, Hanoi.

The climate in TDNP is tropical with two distinct seasons caused by monsoon winds. The hot and rainy season is from May to November, the cold and dry season is from December through April. Average year-round temperature is 23.3°C, with a minimum of 1.8°C and maximum of 41.5°C. The park receives ~160 cm of precipitation annually and relative humidity averages 82%.

TDNP supports at least 1,282 plant species and a large number of animal species, including at least 280 birds, many of which are rare or endangered (Tam Dao National Park 2005). The study area was located on the south-west slope of the Tam Dao range. Sampled forest areas ranged from 200 to 600 m in elevation. The native vegetation in this area can be divided into two types, mature forest and second-growth forest. Mature forest may have had some selective logging in the past, but the forest remains intact with three canopy layers, and with a top canopy height of 30–40 m. Second-growth forest results from intense logging and wood gathering. Small trees comprise second-growth forest vegetation, with forest height usually less than 10 m. The Tam Dao area also contains about 1,500 ha of horsetail pine *Pinus massoniana* plantation, an introduced species. The pine plantations were primarily planted in one main plantation (1,500 ha) with some smaller fragments. The pine plantation is about 30–50 years old.

Avian sampling

Birds were surveyed in the three forest types: mature forest, second-growth forest, and pine plantation. Sampling effort, both in terms of person-hours and area sampled, was approximately the

same in all three forest types to reduce the effects of an area-species richness relationship and to reduce the effects of sampling variability on comparisons of species richness among vegetation types.

All possible 500 m transects were designated on a map of the study area. Transects were chosen to be at least 75 m away from any forest edge to minimise any edge effect. Transects were also separated by at least 100 m to ensure independence among transects. From a random start point, 12 transects were systematically selected in each vegetation type.

Each transect was surveyed six times from June to August 2006. Surveys were carried out under favourable weather conditions. Surveys were conducted from sunrise to noon. Observers walked transects at a constant speed of $\sim$ 0.5 km/40 minutes. While walking transects, observers recorded the species of all birds heard and/or seen except for birds flying overhead which were not recorded. Bird species were classified into two categories, forest specialist or forest generalist, following Cu *et al.* (2000) and Robson (2005).

Data analysis

Many species richness estimators have been developed, almost all of which are preferable to conventional "observed number of species" (Walther and Morand 1998, Walther and Moore 2005). Considerable emphasis has been placed on the use of jackknife estimators (Burnham and Overton 1978) and Chao estimators (Chao 1984, 1987). However, the jackknife and Chao estimators are not based on a maximum-likelihood framework (Walther and Moore 2005); therefore, robust model selection and model uncertainty measurements, including model averaging (Burnham and Anderson 2002), cannot be utilised. In addition, these estimators do not allow the modelling of individual covariates. Individual covariates such as relative abundance, singing propensity, or visual appearance may be expected to explain much of the individual heterogeneity in the detection probability of individual species. More recently, the Huggins estimator (Huggins 1991) has been developed to include individual covariates into the modelling of detection using maximum likelihood theory, but has not been used often in species richness applications (Brand *et al.* 2008). Additionally, Pledger (2000) developed a model that partitions individuals into finite groups (or mixtures) of relatively homogeneous capture probabilities. This model has been used in closed capture-recapture abundance studies (Williams *et al.* 2002), but rarely in species richness applications (e.g. Brand *et al.* 2008, Johnson *et al.* 2009). Heterogeneity in capture (or detection) probability is believed to be more important at a community level (e.g. species-richness; Nichols *et al.* 1998) than at a population level (e.g. abundance) and the heterogeneity in detection probably cannot be explained fully by individual covariates. Because of these advantages of the Huggins and Pledger models, we used an estimator that combines these two models to estimate the number of bird species.

We estimated numbers of forest specialist and forest generalist species in the program MARK (White and Burnham 1999). Data from the 12 transects from one survey period were pooled within vegetation type and survey period and treated as a single sampling occasion (for a total of six sampling occasions within each vegetation type). An encounter history was constructed for each bird species detected during the surveys. Due to sparseness of data and preliminary modelling, two mixtures were used for modelling detection probability with a common probability of inclusion in each mixture across habitat types. Pledger (2000) suggests that using two mixtures is enough to substantially correct for heterogeneity-induced bias in estimation of population size (or in our case, species richness).

Regional commonness index, singing propensity and body length were used as covariates to test predictions concerning detection probabilities. Information used to develop the regional commonness index of each species and information on the body length was inferred from previous avian surveys conducted in TDNP in 2005 (Davidson *et al.* 2005) and from Cu *et al.* (2000) and Robson (2005). Singing propensity was used as an indicator variable in which bird species that can be recognized easily by their typical songs and that sing often (covariate value of 1) were compared with species that are not as easy to detect by song (covariate value of 0). Relationships among body length, singing propensity, and detection probability were assumed to be linear.

We thought the regional commonness index might have a quadratic, rather than a linear, relationship with detection probability because the detection probability does not depend much on abundance if abundance is high.

The importance of these covariates in modelling detection probabilities, as well as vegetation types, was examined using model rankings (ΔAICc), AICc weights (w_i), and cumulative AICc weights (Σw_i) (Burnham and Anderson 2002). Cumulative AICc weight for a given covariate is the sum of the AICc weights of all models that contain that covariate. The main candidate models were: (1) equal detection probabilities (p) in the three vegetation types ($P_{MF=SG=PP}$), where MF = mature forest, SG = second-growth forest, and PP = pine plantation), (2) equal detection probabilities in mature forests and second-growth, with pine plantations being different ($P_{MF=SG\#PP}$); (3) equal detection probabilities in second-growth forest and pine plantations, with mature forest being different ($P_{MF\#SG=PP}$); and (4) different detection probabilities for each of the three vegetation types ($P_{MF\#SG\#PP}$). In addition to vegetation types, we also modelled detection probabilities as a function of survey occasion (t) as an additive (+) and interactive (*) effect. For each of these models we also added covariate effects of observer, singing propensity, body length, and regional commonness index separately or in combination. Models were constructed to estimate number of forest specialist species and number of forest generalist species separately. Parameters of interest were model-averaged across the entire model set. Overall species richness was then estimated by summing the number of forest specialist and forest generalist species and the variance of this estimate was calculated using the Delta method (Powell 2007).

Results

Raw data

Observers recorded 3,648 individual birds and detected 71, 60, and 45 avian species in mature forest, second-growth forest, and pine plantations, respectively. Out of 98 species detected, 49 were classified as forest specialists and 49 as generalists, with 46, 24, and 14 forest specialist species detected in mature forest, second-growth forest, and pine plantation, respectively and 25, 36, and 31 forest generalist species detected in mature forest, second-growth forest, and pine plantation, respectively. These species belonged to eight orders and 21 families. The families Sylviidae, Corvidae, Pycnonotidae, and Nectariniidae were most frequently observed. The most frequently observed species were: Common Tailorbird *Orthotomus sutorius*, Puff-throated Bulbul *Alophoixus pallidus*, Red-whiskered Bulbul *Pycnonotus jocosus*, Grey-cheeked Fulvetta *Alcippe morrisonia*, Striped Tit-babbler *Macronous gularis*, Buff-breasted Babbler *Pellorneum tickelli*, and Puff-throated Babbler *Pellorneum ruficeps*. For the full list of species detected see Vu (2009).

Forest specialist species

Models for forest specialist species in which detection varied by vegetation type had strong explanatory ability, with effects of vegetation type included in all models with $w_i > 0.01$ (Table 1). Models that included the effect of vegetation type were always selected over models that did not include this effect. Detection probability for forest specialist species was highest in mature forest, lower in second-growth forest, and least in pine plantation (Figure 1a). Models with time-varying detection probability did not have much explanatory value (Table 1). All models containing temporal effects had ΔAICc > 5.00 and AICc weight (w_i) < 0.01. Detection probabilities in two mixtures were substantially different (Figure 1a) indicating that 31% of species (95% CI = 18%, 47%) were highly detectable and the rest had low detection probabilities.

Regional commonness index ($\Sigma w_i = 1.00$) had the most influence on detection probability, and consistently appeared in the top models (Table 1 and Figure 2a). Singing propensity ($\Sigma w_i = 0.52$, Figure 2a) and body length ($\Sigma w_i = 0.45$) had similar but weaker influence on detection probability. Models with $w_i > 0.01$ consistently contained the regional commonness index and

Table 1. Model selection results for models describing detection probabilities in three habitats (MF = mature forest, SG = Secondary growth forest, and PP = Pine plantation), with observer (ob), regional commonness (co), singing propensity (si) and body length (bo) effects for forest specialist species with AICc weights ≥ 0.01.

Model	ΔAICc[a]	w_i[b]	K[c]
$P_{(MF\#SG\#PP,}$ co)	0.00	0.10	7
$P_{(MFF\#SG\#PP,}$ co, si, bo)	0.48	0.08	9
$P_{(MFF\#SG\#PP,}$ co, si)	0.77	0.07	8
$P_{(MFF\#SG=PP,}$ co, si, bo)	0.83	0.06	8
$P_{(MF\#SGG=PP,}$ co)	0.91	0.06	6
$P_{(MF\#SGG=PP,}$ co, si)	1.01	0.06	7
$P_{(MF\#SG\#PPPP,}$ ob, co, bo)	1.16	0.05	9
$P_{(MF\#SG\#PPPP,}$ ob, co)	1.23	0.05	8
$P_{(MF\#SG=PP,}$ co, bo)	1.48	0.05	7
$P_{(MF\#SG\#PP,}$ ob, co, si, bo)	1.72	0.04	10
$P_{(MF=SG\#PP,}$ co, bo)	2.00	0.04	7
$P_{(MF\#SG\#PP,}$ ob, co, si)	2.01	0.04	9
$P_{(MF=SG\#PP,}$ co, si)	2.06	0.03	7
$P_{(MF\#SG=PP,}$ ob, co, si, bo)	2.07	0.03	9
$P_{(MF\#SG=PP,}$ ob, co)	2.14	0.03	7
$P_{(MF\#SG=PP,}$ ob, co, si)	2.24	0.03	8
$P_{(MF=SG\#PP,}$ ob, co)	2.29	0.03	7
$P_{(MF=SG\#PP,}$ co, si, bo)	2.69	0.03	8
$P_{(MF\#SG=PP,}$ ob, co, bo)	2.71	0.03	8
$P_{(MF=SG\#PP,}$ ob, co, bo)	3.24	0.02	8
$P_{(MF=SG\#PP,}$ ob, co, si)	3.29	0.02	8
$P_{(MF=SG\#PP,}$ ob, co, si, bo)	3.93	0.01	9

[a]The scaled (minimum AIC = 582.89), small-sample size adjusted Akaike information criterion.
[b]The AICc model weight.
[c]The number of parameters in each model.

models using regional commonness index as a single individual covariate were always selected over models using singing propensity or body length as single individual covariates. Body length only occasionally appeared in the top models.

Based on model-averaged results, we estimated the number of forest specialist species to be highest in mature forest, lower in second-growth forest, and lowest in pine plantations (Figure 3). There was considerable uncertainty in these estimates.

Forest generalist species

Models in which detection varied by vegetation type had strong support; all top models with low ΔAICc values contained effects of vegetation type (Table 2). Models incorporating observer and/or vegetation type effects were always selected over models not incorporating these effects. Detection probability was highest in pine plantation, lower in second-growth forest, and lowest in mature forest (Figure 1b). Models with time-varying detection probability did not have much explanatory value. All models containing temporal effects had ΔAICc > 6.00 and AICc weight (w_i) < 0.02 (Table 2). Detection probabilities in two mixtures were substantially different (Figure 1b) indicating that 26% of species (95% CI = 15%, 41%) were highly detectable and the rest had low detection probabilities.

All three covariates, regional commonness index (Σw_i = 1.00), singing propensity (Σw_i = 1.00), and body length (Σw_i = 0.86) had explanatory ability as some top models included all three of these covariates. Regional commonness index was the best in explaining the variation in detection
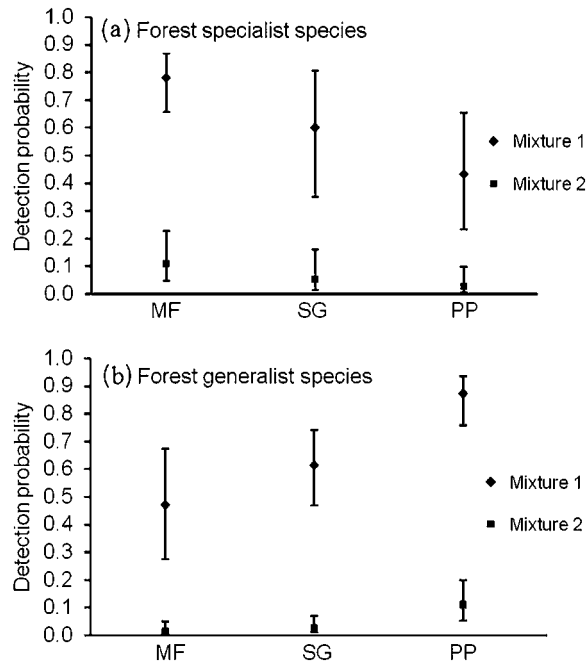
Figure 1. Model-averaged detection probability (with 95% confidence intervals) for the first occasion in the analysis of specialist (a) and generalist (b) species in different vegetation types (MF = mature forest, SG = second-growth forest, and PP = pine plantation). Species with high detection probabilities were categorized in mixture 1 and species with low detection probabilities were categorized in mixture 2.

probability; all models with $w_i > 0.01$ consistently contained regional commonness index (Table 2). Singing propensity consistently appeared in the top models and was the second best explanatory covariate (Table 2 and Figure 2b). Body length had weaker influence on detection probabilities than regional commonness index and singing propensity, and was included less frequently in the top models.

Based on model-averaged results, we estimated number of forest generalist species to be highest in mature forest, lower in second-growth forest and in pine plantation (Figure 3). There was also considerable uncertainty in these estimates.

Total species richness

Using estimates of number of forest specialist and generalist species, we estimated species richness to be highest in mature forest, lower in second-growth forest, and lowest in the pine plantation (Figure 3).

Discussion

Detection probability

Many species richness estimators that incorporate heterogeneity in detection probabilities have been developed and the jackknife (Burnham and Overton 1978) and Chao estimators (Chao 1984, 1987) have received a good deal of attention. However, these estimators do not allow the
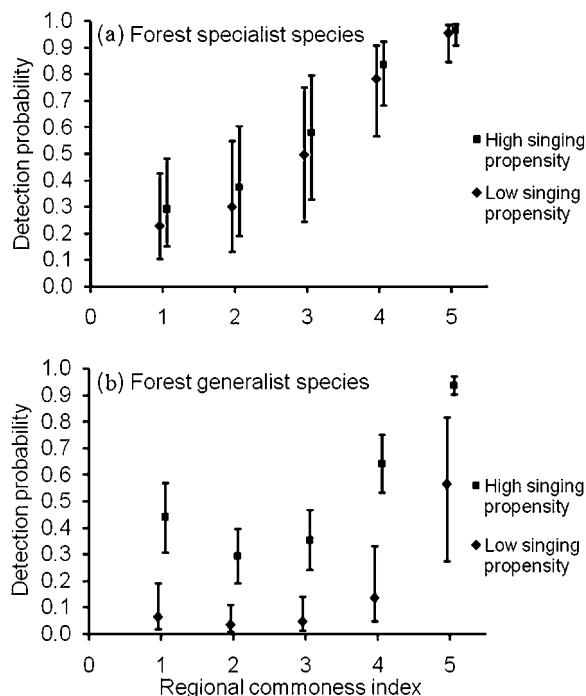
Figure 2. Model-averaged detection probability (with 95% confidence intervals) of species with high and low singing propensity and regional commonness index (1 to 5 with 1 being lowest) during the first occasion for the higher detection probability mixture in mature forest for the forest specialist species (a) and forest generalist species (b).

modelling of individual covariates, and are not based on a maximum-likelihood framework that allows the use of modern model selection criteria (e.g. AIC). Recently, the development of the Huggins (Huggins 1991) and Pledger (2000) estimators have allowed for more flexibility in modelling detection probability by modelling heterogeneity in detection probabilities among

Table 2. Model selection results for models describing detection probabilities in three habitats (MF = mature forest, SG = Secondary growth forest, and PP = Pine plantation), with observer (ob), regional commonness (co), singing propensity (si) and body length (bo) effects for generalist species with AICc weights ≥ 0.01.

Model	ΔAICc[a]	w_i[b]	K[c]
$P_{(MF\#SG\#PP,}$ *ob, co, si, bo)*	0.00	0.51	10
$P_{(MF=SG\#PP,}$ *ob, co, si, bo)*	1.83	0.20	9
$P_{(MF\#SG\#PP,}$ *co, si, bo)*	3.70	0.08	9
$P_{(MF\#SG\#PP,}$ *ob, co, si)*	3.80	0.08	9
$P_{(MF=SG\#PP,}$ *ob, co, si)*	5.05	0.04	8
$P_{(MF=SG\#PP,}$ *co, si, bo)*	5.53	0.03	8
$P_{(MF\#SG\#PP,}$ *+t, co, si, bo)*	6.77	0.02	14
$P_{(MF\#SG\#PP,}$ *co, si)*	7.46	0.01	8
$P_{(MF=SG\#PP,}$ *+t, co, si, bo)*	8.56	0.01	13
$P_{(MF=SG\#PP,}$ *co, si)*	8.70	0.01	7

[a]The scaled (minimum AIC = 582.89), small-sample size adjusted Akaike information criterion.
[b]The AICc model weight.
[c]The number of parameters in each model.