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Do landscape features predict the presence of barn owls in a changing agricultural landscape?

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HIGHLIGHTS

- ► We investigated how changes to the agricultural landscape influences nest site use by barn owls.
- ► Thirty percent of nest sites used in the early 1990s were lost by 2007.
- ► Loss of grassland habitat did not predict the continued use of remaining nest sites.
- ► Barn owls were less likely to persist at sites with increased highway traffic exposure.
- ► The length of highway within a 1-km radius of available nest sites influences whether they are occupied by barn owls.

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ABSTRACT

Population declines of farmland birds have been linked to the loss and fragmentation of grassland habitats resulting from changes in agricultural practices and urbanization. We investigated how changes to landscape attributes in the Fraser Valley, British Columbia, Canada, influenced the persistence and current occupancy by barn owls at roosting and nesting sites. There has been considerable development in the agricultural landscape of the Fraser Valley between the early 1990's and 2007/2008: grassland cover declined by 53%, the area of urban cover increased by 133%, length of secondary roads increased by 18%, and the volume of highway traffic increased by 33%. We also found that 30% of the sites used by barn owls in the early 1990s have been lost. Although the availability of grasslands are thought to influence the distribution of barn owls, in our study, barn owls were not more likely to persist at sites with little loss of grass cover, or to currently occupy sites surrounded by more grassland. The only variables that predicted the continued use and current occupancy of sites were traffic exposure and the length of highways. Barn owls were most likely to persist at sites with lower increases in traffic exposure and occupied sites containing fewer kilometers of highway within a 1-km radius. We conclude that the distribution of barn owls in the Fraser Valley is influenced by the loss of suitable roosting and nesting sites and location of highways.

1. Introduction

Agricultural landscapes provide important habitat for many plant and animal species. Traditional farming practices such as crop rotations and the maintenance of hedgerow and grassy verges produce a structurally varied landscape capable of maintaining levels of biodiversity similar to that of many natural ecosystems (Bignal & McCracken, 1996; Bohlen & House, 2009). However, changes in agricultural practices over the last 50 years (e.g. use of agrochemicals, modernized machinery) have reduced the heterogeneity of agricultural landscapes, resulting in large, heavily utilized, monoculture fields and an overall reduction in the quality of land as habitat for wildlife.

In addition to changes in agricultural practices, in many areas agricultural land has been lost and fragmented due to urbanization and its associated infrastructure (Forman et al., 2003; Underhill & Angold, 2000). Although urbanization results in habitat loss, infrastructure such as roads and railways can have a disproportionate impact on wildlife due to mortality from collisions with vehicles, restricted access to resources through the barrier effect, or because populations become subdivided and isolated into smaller and more vulnerable fractions (Forman et al., 2003; Jaeger et al., 2005). These changes have been implicated in the declines both in

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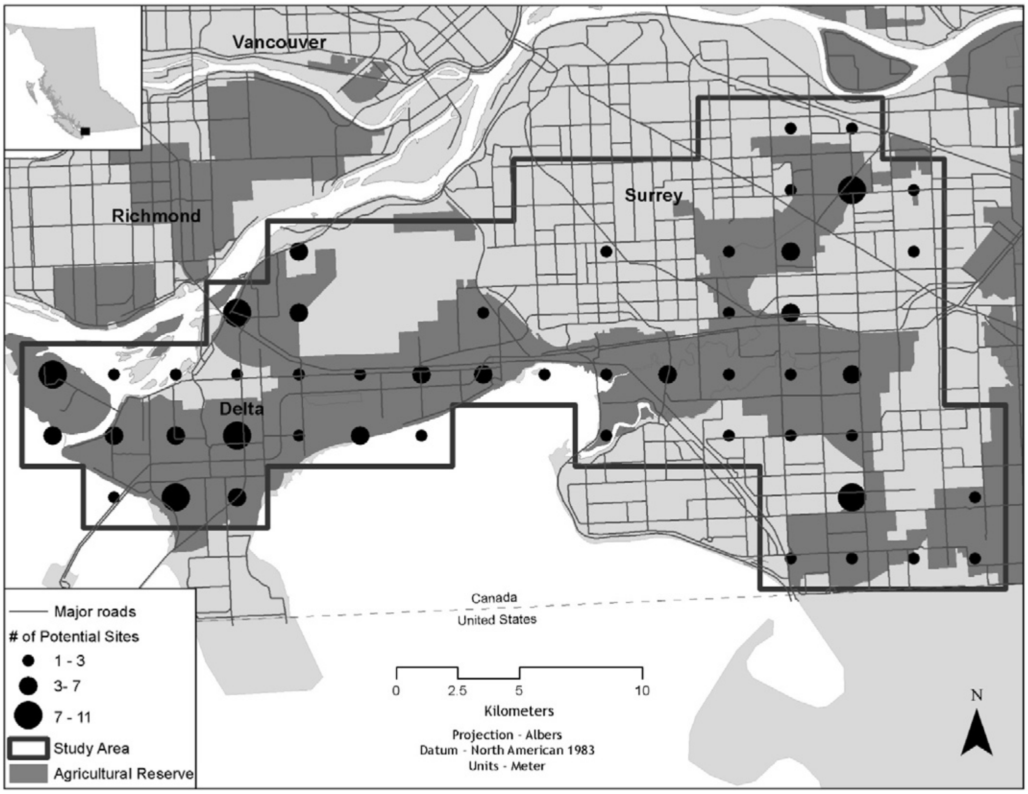

Fig. 1. Study area within the Fraser Valley, British Columbia, Canada used for assessing landscape features that influence nest-site use by barn owls in 2007 and 2008. Exact locations of available nest sites are not indicated because barn owls are a species listed under the Species at Risk Act in Canada and most sites are located on private land.

range and abundance of many species associated with agricultural landscapes (Benton, Vickery, & Wilson, 2003; Filippi-Codaccioni, Devictor, Clobert, & Julliard, 2008; Robinson & Sutherland, 2002).

There is accumulating evidence that farmland birds as a group are particularly sensitive to changes to the agricultural landscape. In Britain, 86% of farmland birds have shown range contractions over the last 40 years (Fuller et al., 1995). Similar trends have been reported from elsewhere in Europe and North America (Brennan & Kuvlesky, 2005; Donald, Sanderson, Burfield, & van Bommel, 2006; Peterjohn, 2003). However, no single factor appears to be responsible for the observed range contractions and population declines (Newton, 2004). For example, population declines in corn bunting (*Miliaria calandra*) and tree sparrow (*Passer montanus*) have been linked to a reduction in winter survival due to loss of fallow grain fields, a key winter food supply (Siriwardena, Robinson, & Crick, 2002; Wilson, Boyle, Jackson, Lowe, & Wilkinson, 2007). In contrast, population declines in lapwing (*Vanellus vanellus*) appear to be linked to higher rates of nest predation resulting from increased grazing intensity on marginal grasslands (Chamberlain & Crick, 2003). The variation in underlying causes for population declines of farmland birds has led some authors to argue that more species-specific research is required in order to design appropriate conservation measures (Fuller et al., 1995; Peterjohn, 2003).

The barn owl (*Tyto alba*) is an iconic farmland bird that has been associated with humans and agriculture for centuries (Bunn, Warburton, & Wilson, 1982). However, barn owls are now experiencing range contractions and declines across their range (Colvin, 1985; Toms, Crick, & Shawyer, 2001). In Britain, where they have been studied most intensively, their numbers are reported to have dropped by 69% over the last 50 years (Toms et al., 2001). Three major factors have been associated with declines in barn owl population. First, the loss of moderate-length grassland over the last 50 years has decreased small mammal populations and potentially prey availability (Colvin, 1985; Taylor, 1994). Second, suitable nest/roosting sites have been lost due to the conversion of old wooden barns into inaccessible steel barns and the removal of old trees as part of field enlargement programs (Ramsden, 1998; Taylor, 1994). Third, adult mortality may have increased due to increases in the number of roads and traffic volume in agricultural areas (Preston & Powers, 2006; Ramsden, 2003).

Population trends for barn owls are not well documented in North America but, based on breeding bird surveys the barn owl is thought to be declining by up to 3% per year in western North America (Environment Canada, 2010). The barn owl population in western Canada was designated as “Threatened” by the Committee on the Status of Endangered Wildlife in Canada in 2010 (COSEWIC, 2010). To evaluate how changes to the agricultural landscape have impacted the distribution of barn owls in Western Canada, we examine how changes in landscape features over the past 15 years have impacted occupancy at historical nest/roost sites, and examine how current landscape features influence the use of suitable nest/roost sites across the Fraser Valley, British Columbia. Specifically we use an information theoretic approach to assess the role of four features of the landscape surrounding available nest sites hypothesized to impact the distribution and breeding of barn owls: (i) the amount of grassland cover (Colvin, 1985; Taylor, 1994), (ii) the fragmentation of grassland habitat (Taylor, 1994), (iii) urban development and (iv) traffic exposure, or length of highways (Bond, Burnside, Metcalfe, Scott, & Blamire, 2005; Lodé, 2000; Ramsden, 2003).

2. Methods

2.1. Study site

We monitored barn owls in the municipalities of Delta and Surrey, an area of 681 km^2 within the Fraser Valley, British Colombia, Canada (49°8′0″ North, 122°18′0″ West; Fig. 1). This area is of key importance for migratory and residential farmland birds and encompasses important wildlife areas such as the Alaksen National Wildlife Area, Burns Bog and Boundary Bay. Prior to European settlement the floodplain would have been dominated by grassland and low shrub vegetation while higher elevations would have been covered primarily by coniferous forest (North & Teversham, 1983). Since European settlement the area has traditionally been used for pasture and hay production, but over the last 35 years agricultural practices have shifted to vegetable and berry production in fields and greenhouses (Statistics Canada, 2006). The current population of Delta and Surrey combined is seven times larger than the 1960s population. This has coincided with the removal of ~5000 ha (7.5%) of land from agricultural production in this area since 1974 (Agricultural Land Reserve, 2009).

2.2. Survey methodology

To evaluate changes in the availability and occupancy of barn owl nesting/roosting sites in Delta and Surrey we revisited 116 of the 129 sites occupied by barn owls in this area in the 1990s (Andrusiak, 1994). We were unable to obtain the landowners permission to access the remaining 13 sites. In total, 80 of these sites remained available for barn owls (see results). To evaluate the current distribution of barn owls we surveyed all potentially suitable nesting and roosting sites that we could access in Delta and Surrey ($n = 143$ sites). This sample included the 80 sites surveyed by Andrusiak (1994) that were still available for barn owls and an additional 63 sites not surveyed in the 1990s. Potentially suitable sites included old wooden barns or other tall structures with suitable openings near the roof and old single-standing trees on farm properties (Bunn et al., 1982; Taylor, 1994).

Sites were monitored for seven months over the peak barn owl breeding period in BC for two years (March to September 2007 and March to September 2008; Campbell et al., 1990). To ensure that occupancy was not underestimated, we surveyed all potential sites at least three times per year. During each survey we searched the inside and perimeter of the structure for barn owls or indications of their presence (such as fresh pellets, feathers and/or prey carcasses). We also climbed up the side of the structure to check beams/platforms for any nesting activity. If the beams/platforms could not be accessed at a site, a 30 min. observation was conducted at dusk, to determine if any barn owls exited or entered the structure. A structure was considered occupied if a barn owl was sighted or if we found fresh pellets or feathers during any of the three surveys. Interviews with landowners confirmed that all barns where no barn owls were sighted and no fresh pellets or feathers were found were not currently being used by barn owls. We subsequently visited all occupied sites every 1–2 months to determine if they were used for breeding. Visits of this frequency enable evaluation of whether breeding occurred, as incubation and rearing takes approximately 2.5 months, and females and chicks beg loudly during the incubation and nestling periods. We could not determine the breeding status at all occupied sites due to limitations on our ability to access and search sites; breeding status was determined for 106 of the 115 occupied sites.

2.3. Land use characteristics and spatial analysis

We quantified current and historical land use within a 1-km radius of each potential nesting/roosting site from digitized data layers using Geographic Information System software (ArcGIS 9.2). We used a 1-km radius around each site because this results in an area that approximates the home range of a barn owl (3 km^2; Bond et al., 2005; Shawyer & Shawyer, 1995; Taylor, 1994).

Grassland and crop cover in Delta was obtained from an individual field layer from 2007 that contained information on grassland and crop type (Ducks Unlimited, unpublished data). A similar data layer was not available for Surrey, so we created a data layer by visually inspecting individual fields for grassland or crop type and digitizing these data ($n = 1747$ fields). Data on urban cover (housing, commercial and industrial land use) were obtained from a 2006 Vancouver Regional District land-use layer map for the entire study area (Metro Vancouver, 2008). Data on highways, connecting ramps, and roads within the study area were obtained from a BC road layer map (GeoBC, 2010).

We categorized grass cover into five different categories based on the intensity of use, a factor linked to vole densities (Aschwanden, Holzgang, & Jenni, 2007; Butet & Leroux, 2001; Tattersall, Avundo, Manley, Hart, & MacDonald, 2000): (i) permanent grasslands, (ii) grassland set asides (areas of grass that are not disturbed for up to four years), (iii) pastures (areas of grass that are lightly grazed), (iv) hayfields, areas of hay crop that is mowed up to four times/year, (v) grassy verges along roads and field edges (mowed 4–6 times/year). We further categorized grass cover into high quality foraging habitat that is not mowed (permanent grasslands, grassland set asides and pasture) versus lower quality habitat that is regularly mowed (hayfields, grass verges, field edges). Grassland types did not change between 2007 and 2008. Crop types in fields varied between years but crop cover was not used as an explanatory variable in analyses.

To reconstruct land use in the study area during the 1990s and create data layers of historic land use we integrated information from several sources. We identified grassland and crop cover in the 1990s from high-resolution ortho photos taken in May 1995 (1 m pixel). Grass cover can be distinguished easily from other crops in these photos based on color and texture ($n = 1801$ fields). We were unable to distinguish between high quality and other foraging habitat in these photos. Data on urban cover in the 1990s was obtained from the Vancouver Regional District 1996 land-use map (Metro Vancouver, 2008). Data on highways, connecting ramps, and roads within the study area during the 1990s were obtained from a BC road layer map (British Columbia Ministry of Transportation, 2008). No additional highways were built in Delta or Surrey between 1990 and 2008, although, some "secondary roads" were upgraded to "highways". To assess the impact of highways on the distribution of barn owls over this period we therefore calculated the change in the exposure to highway traffic at each of the historical sites that remained suitable for barn owls. We did this by multiplying the length (km) of highway and connecting ramps within 1 km of each site by the historical (1990–1993) and current (2007) average traffic volumes for the appropriate section of the highway. Data on average traffic volumes on highways for 1990 and 2007 were obtained from the Ministry of Transport (British Columbia Ministry of Transportation, 2008). Average traffic volume is defined as the average number of vehicles passing a fixed point on a highway/highway ramp over the course of a day.

2.4. Data analysis

We characterized land use change within a 1-km radius of each site surveyed in 1990–1992 that remained suitable for barn owls using the following explanatory variables: (i) change in area of grass cover, (ii) change in number of grass patches, (iii) change in average distance to grass patches, (iv) change in area of urban cover, (v) change in the length of secondary roads, and (vi) change in highway traffic exposure. We log-transformed three variables (change in urban cover, change in secondary roads and change in highway traffic exposure) to reduce skew. Log-transformed change in urban cover was highly correlated with log-transformed change in length of secondary roads (Pearson's correlation, $r_p = 0.65$, $n = 80$, $p < 0.001$) so we excluded the change in secondary road variable from subsequent analyses. There was some collinearity among the remaining variables (Pearson's correlation, r_p, range −0.33 to +0.25, $n = 80$), but all variables had variance inflation factors that were close to 1 (range 1.04–2.22) and stable regression coefficients indicating multicollinearity was not a cause for concern (Menard, 1995). We then developed a candidate model set based on the hypothesized effects of foraging habitat loss, foraging habitat fragmentation, urban development and highway traffic exposure on the continued use of suitable nest sites by barn owls. The influence of foraging habitat loss was evaluated using the change in area of grass cover, foraging habitat fragmentation using change in number of grass patches and/or change in the average distance to grass patches, urban development using change in area of urban cover, and level of traffic exposure using change in highway traffic exposure. The candidate model set evaluated the independent effects of habitat loss (1 model), fragmentation (3 models; number of grass patches, average distance to grass patches, number of grass patches + average distance to grass patches), urban development (1 model) and traffic exposure (1 model), the combined effects of habitat loss and fragmentation (1 model), the combined effects of urban development and traffic exposure (1 model), the combined effect of all four factors (1 model) and a null intercept-only model ($n = 10$ models). Models were fitted using a Generalized Linear Model with a binary logistic structure and a logit link.

We next examined whether current occupancy status at a site (occupied/unoccupied $n = 143$ sites) and breeding status at a site (breeding/nonbreeding, $n = 106$ occupied sites) were influenced by the current land use surrounding a site. We characterized land use within a 1-km radius of each site using the following explanatory variables: (i) total area of grass cover, (ii) area of high quality grass cover, (iii) number of grass patches, (iv) average distance to grass patches, (v) area of urban cover (impermeable surface = housing, industrial, commercial and greenhouses), (vi) length of secondary roads, and (vii) length of highway. We log-transformed variables where appropriate to reduce skew. In the larger dataset, log-transformed urban cover was highly correlated with log-transformed change in length of secondary roads (Pearson's correlation, $r_p = 0.59$, $n = 143$, $p < 0.001$) so we excluded the secondary road variable from subsequent analyses. The two grass cover variables were also correlated (Pearson's correlation, $r_p = 0.35$, $n = 143$, $p < 0.001$) but these variables were not included in the same model (see below). There was some collinearity among the remaining variables (Pearson's correlation, r_p, range, −0.16 to +0.33, $n = 143$), but all variables had low variance inflation factors (range 1.04–2.69) and stable regression coefficients indicating multicollinearity was not a cause for concern (Menard, 1995). We then developed a candidate model set based on the hypothesized effects of foraging habitat availability, foraging habitat fragmentation, urban development and highways on the use of suitable nest sites by barn owls. The influence of foraging habitat availability was evaluated using the total area of grass cover or the area of high quality grass cover, foraging habitat fragmentation using the number of grass patches and/or the average distance to grass patches, urban development using the area of urban cover, and highways using the length of highway. The candidate model set evaluated the independent effects of foraging habitat availability (2 models; total grass cover, high quality grass cover), fragmentation (3 models, number of grass patches, average distance to grass patches, number of grass patches + average distance to grass patches), urban development (1 model) and highways (1 model), the combined effects of foraging habitat availability and fragmentation (2 models; total grass cover + number of grass patches + average distance to grass patches, high quality grass cover + number of grass patches + average distance to grass patches), the combined effects of urban development and traffic exposure (1 model), the combined effect of all four factors (2 models; with either the total grass cover or high quality grass cover variable) and a null intercept-only model ($n = 13$ models). Models were fitted using a Generalized Linear Model with a binary logistic structure and a logit link.

We assessed whether spatial autocorrelation could cause us to overestimate the importance of landscape variables in our analyses using Moran's tests and visual examination of Moran's I correlograms and semivariograms of the residuals with 95% confidence envelopes (SAS 9.2, proc variogram; Fortin & Dale, 2005). We detected no evidence of spatial autocorrelation in the model residuals (Moran's I, all $p > 0.90$). In all models, Moran's I was close to zero for all separation distances and semivariance did not increase with lag distance, suggesting little spatial autocorrelation.

We used an information theoretic approach to rank and identify the best-supported models within each model set (Burnham & Anderson, 2002). We calculated Akaike's Information Criterion adjusted for small sample sizes (AICc) and AICc weights (w_i) for each model. Models with AICc scores within 2 of the best model and with high w_i values were considered to have strong support (Burnham & Anderson, 2002). We calculated the weighted parameter estimates and their unconditional standard errors using all models in the candidate set (Burnham & Anderson, 2002). We also calculated Nagelkerke's r^2 to evaluate the amount of variation explained by each model (Nagelkerke, 1991). All data were analyzed using SPSS version 16.0 (SPSS Inc.; Chicago, Illinois).

3. Results

Land use within a 1-km radius of each potential barn owl site changed considerably between the 1990s and 2007/2008. Grass cover around sites decreased by 53%, whereas all measures of development increased: the area of urban cover increased by 133%, the length of secondary roads increased by 18%, and traffic exposure increased by 33% (Table 1). The length of highway only increased at three of the 80 sites (Table 1)

We found in 2007/2008 that barn owls continued to occupy 68 of the 116 sites (58.6%) that were occupied in the Fraser Valley in the early 1990s. However, many of the previously occupied sites were no longer available because barns or roost trees had been removed ($n = 26$) or old wooden barns had been replaced by steel barns that were inaccessible to barn owls ($n = 10$). Overall, nearly one-third of sites occupied 15 years ago were no longer available to barn owls. After excluding sites that had been lost, 84% (68 of 80) of the sites occupied in the 1990s remained occupied in 2007/2008.

3.1. Continued occupancy of nesting/roosting sites used in the 1990s

Three of the ten models examining how changes in land use impact the continued use of sites occupied by barn owls in the 1990s received strong support ($\Delta AIC_C < 2$; Table 2a). The highest ranked model which received 2.6 times the support of the null model included the change in traffic exposure term. The change in traffic exposure term was also included in the next best model and models including this term received 56% of the AIC weight. Barn owls were less likely to continue to occupy sites with increased traffic exposure (Table 3). The probability that sites remained occupied was approximately 90% if there was little or no change in traffic exposure but decreased to 60% at sites with the greatest increase in highway traffic exposure (Fig. 2).

Table 1
Summary of changes in land use within a 1-km radius of a suitable barn owl roosting or nesting sites comparing all sites that were surveyed in the 1990s and in 2007/2008 ($n = 80$). Means ± SD presented for each variable.

Variable	Mean 1990s	Mean 2007/2008	Wilcoxon
Grass cover (km^2)	0.75 ± 0.37	0.48 ± 0.38	$Z = -6.37$, $p < 0.001$
Number of grass patches	6.8 ± 2.7	5.8 ± 2.6	$Z = -3.57$, $p < 0.001$
Average distance to grass patches (m)	584 ± 66	559 ± 100	$Z = -2.19$, $p = 0.028$
Urban cover (km^2)	0.18 ± 0.41	0.41 ± 0.52	$Z = 6.67$, $p < 0.001$
Length of highway (km)	1.64 ± 2.16	1.74 ± 2.27	$Z = 1.60$, $p = 0.11$
Length of secondary roads (km)	7.45 ± 4.90	8.79 ± 5.27	$Z = 3.43$, $p = 0.001$
Highway traffic exposure (number of vehicles × 10^{-3} km/day)	30.60 ± 49.18	40.10 ± 62.33	$Z = 5.37$, $p < 0.001$

Table 2
Summary of AIC models examining the relationship between (a) changes in land use and the continued occupancy of barn owl nesting/roosting sites used in the 1990s, (b) land use and the occupancy of potential nesting/roosting sites in 2007/2008, and (c) land use and breeding at occupied sites in 2007/2008. The results for all strongly supported models (ΔAICc < 2.0) and the null model are presented (K: number of parameters estimated, N: sample size, ΔAICc: the differences between the AICc of each model and the model with the highest AICc score, w_i: AIC weight for that model, Evidence ratio: $w_i/w_{i\ \text{best model}}$, r^2: Nagelkerke's pseudo r^2).

Model	K	N	AICc	ΔAICc	w_i	Evidence ratio	r^2
(a)							
ΔHighway traffic exposure	2	80	66.48	0.00	0.31	1.00	0.11
ΔHighway traffic exposure + Δurban cover	3	80	67.89	1.41	0.15	0.48	0.13
ΔDistance to grass patches	2	80	68.00	1.52	0.15	0.48	0.08
Null	1	80	69.69	3.21	0.12	0.39	0.00
(b)							
Length of highway	2	143	141.63	0.00	0.28	1.00	0.04
Total grass cover	2	143	143.05	1.42	0.14	0.50	0.03
Null	1	143	143.46	1.83	0.11	0.39	0.00
(c)							
Null	1	106	124.44	0	0.25	1.00	0.00
Total grass cover	2	106	125.89	1.45	0.12	0.48	0.01
Number of grass patches	2	106	126.10	1.66	0.11	0.44	0.006
Length of highway	2	106	126.17	1.73	0.11	0.44	0.005
Urban cover	2	106	126.35	1.91	0.10	0.40	0.002
Distance to grass patches	2	106	126.39	1.95	0.10	0.40	0.002

Changes in urban cover and changes in the average distance to grass patches were each included in one of the strongly supported models (Table 2a). However, models with these variables had lower summed AIC weights, and the variables had weighted parameter estimates that were smaller than their unconditional standard errors (Table 3). Changes in grass cover and changes in the number of grass patches were not included in any of the strongly supported models. Consequently there was little evidence that changes in grass cover, the number and distance to grass patches or the amount of urban cover influenced the continued use of sites occupied by barn owls in the 1990s.

Table 3
Summed AIC weights for models and weighted parameter estimates and unconditional standard errors for variables included in the candidate models examining the relationship between (a) changes in land use and the continued occupancy of barn owl nesting/roosting sites used in the 1990s, (b) land use and the occupancy of potential nesting/roosting sites in 2007/2008.

Variable	Σw_i	Weighted parameter estimate	Unconditional SE
Intercept		2.147	0.603
ΔHighway traffic exposure	0.56	−0.684	0.424
ΔGrass cover	0.17	−0.014	0.244
ΔDistance to grass patches	0.39	−0.003	0.002
ΔNumber of grass patches	0.29	0.054	0.060
ΔUrban cover	0.28	1.045	1.808
Intercept		1.437	0.614
Length of highways	0.42	−0.535	0.410
Total grass cover	0.19	0.517	0.605
High quality grass cover	0.12	0.651	0.812
Distance to grass patches	0.17	0.000	0.000
Number of grass patches	0.19	0.027	0.032
Urban cover	0.17	0.018	0.432

3.2. Current occupancy of potential nesting/roosting sites

Barn owls occupied 115 of the 143 (80.4%), potentially suitable sites surveyed in the Fraser Valley in 2007 and 2008. Occupied sites contained more grass cover, less urban cover and fewer highways and roads relative to unoccupied sites (Table 4). Three of the 13 models examining hypotheses for how current land use influences barn owl occupancy received strong support ($\Delta AIC_C < 2$; Table 2b). The top ranked model which received 2.5 times the support of the

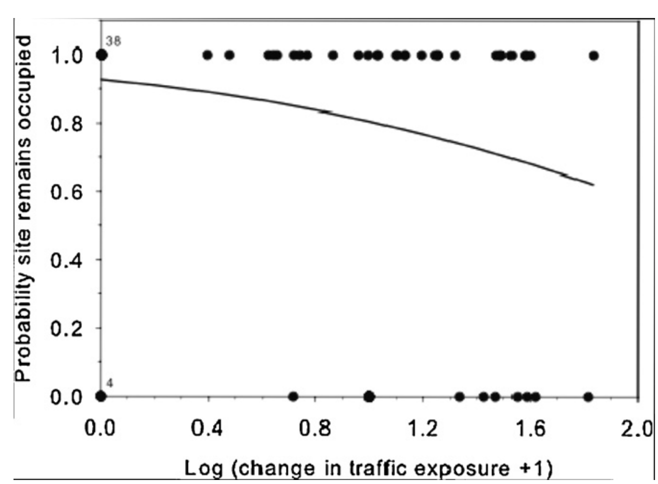

Fig. 2. The relationship between the change in traffic exposure (log-transformed) and the probability sites occupied by barn owls in the 1990s remain occupied in 2007 and 2008. Traffic exposure is the average traffic volume × 10^{-3} per day multiplied by the length of highway within a 1-km radius of each site. The points show the data and the numbers indicate the sample size for sites where there was no change in traffic exposure. The line shows the prediction from the best supported model in the candidate model set.

Table 4
Summary of current land use within a 1-km radius of occupied and unoccupied sites in 2007/2008 (n = 143). Means ± SD are presented.

Variable	Occupied (115)	Unoccupied (28)
Total grass cover (km^2)	0.50 ± 0.40	0.38 ± 0.28
High quality grass cover (km^2)	0.11 ± 0.14	0.08 ± 0.11
Number of grass patches	5.2 ± 2.4	4.8 ± 1.6
Average distance to grass patches (m)	537 ± 98	551 ± 143
Urban cover (km^2)	0.38 ± 0.48	0.39 ± 0.46
Length of highways (km)	1.36 ± 1.98	2.53 ± 2.76
Length of secondary roads (km)	6.49 ± 4.50	6.62 ± 3.69

null model included the length of highway term. Models including this term accounted for 40% of the AIC weight (Table 3). The likelihood that an apparently suitable site was occupied by a barn owl decreased as the length of highway increased. Approximately 85% of sites were occupied if there was less than one km of highway within the home range, whereas approximately 65% of sites were occupied if there was more than seven km of highways within the home range (Fig. 3).

The total grass cover term was included in one of the supported models (Table 2b). However, models with this term had a low summed AIC weight, and the weighted parameter estimates for this variable had a standard error that bounded zero (Table 3). The high quality grass term, the number or average distance to grass patches terms, and the urban cover term were not included in any of the strongly supported models. We conclude that the area of grassland, the number and location of grass patches and the area of urban cover within 1 km of a nest/roosting site had little influence on whether sites were occupied by barn owls.

3.3. Breeding status

Barn owls bred in 78 of the 106 (73%) occupied sites. However, we found little evidence that any of the landscape variables measured discriminated between the sites that were or were not used for breeding. Although six of the 13 models received strong AIC support, the top ranked model was the null model. The null model received more than twice the support of any model including a landscape variable (Table 2c). We conclude that none of the landscape variables assessed predicted breeding by barn owls at occupied sites.

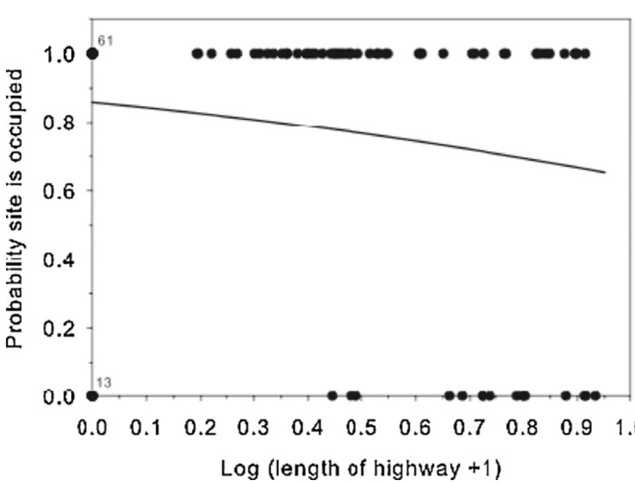

Fig. 3. The relationship between the length of highway within a 1-km radius of each potential nest/roost site (log-transformed) and the probability that the site is occupied by a barn owl. The points show the data and the numbers indicate the sample size for sites where there is no highway within a 1-km radius of the site. The line shows the predicted values from the best supported model in the candidate model set.

4. Discussion

Previous research suggests that barn owl populations are vulnerable to two major factors: loss of suitable nesting sites and the loss of grassland areas for foraging (Colvin, 1985; Taylor, 1994). Both suitable nesting sites and foraging habitat have been lost in the Fraser Valley over the last 15 years: almost one-third of suitable breeding sites for barn owls have been removed, grass cover in the study area has decreased by more than half and urban cover has increased. Despite these changes, barn owls did not occupy all apparently suitable sites and the proportion of occupied sites remained similar in both periods. Sites with lower proximity to or availability of grasslands were not less likely to remain occupied, or to be currently occupied. The only landscape variables that consistently predicted the continued use and current occupancy of sites by barn owls were changes in the volume of highway traffic, and the length of highways within a 1-km radius of a potential roosting or nesting site. This suggests that major roads can have an impact on the distribution of barn owls in agricultural landscapes.

The declines in populations of barn owls in Western Europe and North America are associated with the ongoing loss of suitable nests sites associated with urbanization and changes to agricultural practices (Bunn et al., 1982; Percival, 1991; Taylor, 1994). In southern Scotland an estimated 4–6% of suitable nest sites were lost per year over a 13-year period (Taylor, 1994). Nest sites are also being lost in the Fraser Valley. Within our study area, we estimate that suitable sites are being lost at ~2.2% per year over the last 15 years. Nest sites were lost mainly due to urban development, but also due to changes in barn design, making barns inaccessible to owls. In addition, the availability of natural nest sites is decreasing; all roosting sites in trees occupied in the 1990s had been lost by 2007/2008. The number of large single-standing trees has been declining in many agricultural landscapes over the last 50 years due to field enlargement programs associated with agricultural intensification (Taylor, 1994).

Shortages of suitable nest sites may limit some barn owl populations. For example, in southern Scotland the density of breeding barn owls increased on plots containing nest boxes (Shawyer, 1998). Nest boxes were also quickly occupied in Northern Utah: 80% of boxes were found to contain breeding pairs of barn owls two years after deployment (Marti, Wagner, & Denne, 1979). We found that a high proportion of apparently suitable nest sites were occupied by barn owls. Farmers in the area also report that barn owls rapidly occupy and breed in newly installed nest boxes. Nest site availability is therefore likely to play a role in the distribution of barn owls within agricultural areas within the Fraser Valley.

Changes in the configuration of the agricultural landscape, especially increased fragmentation owing to road development, negatively impacts many species of wildlife (Fahrig & Rytwinski, 2009; Forman et al., 2003; Jaeger et al., 2005). We found that highway traffic and the location of highways has some influence on the distribution of barn owls in the Fraser Valley. Highway traffic exposure and the length of highways could influence nest/roost site occupancy in barn owls because barn owls using sites in proximity to highways suffer higher mortality, barn owls avoid areas where noise and disturbance reduce foraging habitat quality or foraging efficiency, or traffic noise and disturbance influences nest site selection. Vehicle collisions can be a major cause of barn owl mortality in Europe and North America (Baudvin, 1997; Boves, 2007; Fajardo, 2001; Lodé, 2000; Newton, Wyllie, & Asher, 1991; Preston & Powers, 2006; Ramsden, 2003). Mortality rates are particularly high on highways that are elevated compared to the rest of the surrounding landscape (Baudvin, 1997; Lodé, 2000). Vehicle collisions are known to kill and injure a large number of owls within our study area (Preston & Powers, 2006). For example, Andrusiak (1994) reported that 63% (n = 341) of the barn owls found dead in the Fraser

Valley were killed by collisions with vehicles. However, we have no direct evidence linking changes in site occupancy or the absence of barn owls at suitable sites with adult mortality on highways. Traffic noise has been shown to reduce the hunting efficiency of acoustic predators such as owls and bats (Barber, Crooks, & Fristrup, 2010; Siemers & Schaub, 2011). However, we and others (e.g. Lodé, 2000; Ramsden, 2003) frequently observed owls hunting on the grassy highway verges suggesting that barn owls do not avoid these habitats due to noise or disturbance. There is also little evidence that highway traffic reduces habitat quality for barn owls; McGregor, Bender, and Fahrig (2008) found that the density of rodents was not reduced in suitable habitat around roads. Anthropogenic noise has also been shown to influence site occupancy by birds, reducing overall species richness and the breeding density of some species at affected sites (Francis, Ortega, & Cruz, 2009; Reijnen, Foppen, ter Braak, & Thissen, 1995). Data on the survival of marked individuals and recruitment at suitable nest sites that differ in highway traffic exposure is required to determine the mechanistic basis for the patterns observed in our study.

Barn owls, as vole specialists (*Microtus* spp. typically contribute 50–70% of their diet; Marti, Poole, & Bevier, 2005; Taylor, 1994) should be strongly impacted by changes in the quality and quantity of grasslands that they use for foraging. Loss of high quality grassland habitats is linked to overall declines in vole populations in agricultural landscapes across Europe (Butet & Leroux, 2001). In addition, other studies have found a strong relationship between the area of non-intensively used grassland and the density of voles (Aschwanden et al., 2007; Gorman & Reynolds, 1993). Barn owls, in southern England, were estimated to require 0.40 km^2 of unimproved grassland within their home range (Shawyer & Shawyer, 1995). Barn owls in the Fraser Valley currently have less high quality grassland within their home range (0.11 km^2).

Surprisingly, however, we found no evidence that barn owls were more likely to continue to occupy sites that maintained areas of grassland, or more likely to be found at sites surrounded by more high quality, or total, grass cover. Site occupancy by barn owls within our study site may not have been impacted by loss and availability of high quality foraging habitat because they compensate by increasing their foraging in lower quality grass habitats within their home range. Some apparently lower quality grass habitats, such as the verges alongside fields, roads and highways, can also have high densities of voles (e.g. field edges, Taylor, 1994; and roadside verges, Bolger, Scott, & Rotenberry, 2001). In agricultural landscapes where little high quality undisturbed permanent grassland habitat remains, grassy verges along roads and fields are likely to become increasingly important, but potentially risky, foraging habitats for barn owls.

Alternatively, site occupancy by barn owls may not be linked to the availability of grassland foraging habitats if barn owls can shift to alternate prey and utilize other habitat types. Although barn owls are vole specialists (Derting & Cranford, 1989; Fast & Ambrose, 1976) the proportion of voles in their diet can vary widely. For example, many previous studies have shown prey shifting by barn owls in response to temporal changes in vole abundance (Colvin, 1984; Webster, 1973). Barn owls residing in urban areas have diets containing a higher proportion (13–55%) of rats and birds (Buckley & Goldsmith, 1975; Campbell et al., 1990; Salvati, Ranazzi, & Manganaro, 2002). These patterns were also apparent within our study; pellet analysis showed that barn owls at different nest sites varied in their dependence on voles (*Microtus* spp.; Hindmarch, 2010).

Proximity to suitable foraging habitat should be most important when individuals are breeding, and therefore constrained to return to the nest site. For example, Bond et al. (2005) found that barn owls were more likely to successfully breed at sites adjacent to higher-quality small mammal habitat with lower overall habitat heterogeneity. In contrast, we could find no evidence that the amount or distance to grass cover influenced whether a site was used for breeding. However, we were unable to restrict our analyses to sites occupied by females, potentially obscuring any pattern. Alternatively, differences in results between our study and Bond et al.'s may have been due to slight analytical differences: they examined breeding success across all possible sites, whereas we examined the probability of breeding only at occupied sites. We are currently examining the relationship between barn owl productivity and landscape features.

The landscape features measured in this study were weak predictors of site occupancy by barn owls. The low predictive ability of the landscape variables is likely due, in part, to the high rates of continued use and current occupancy of available nest sites but also suggests that other factors play a role in nest/roosting site use by barn owls. Specific features of the structure and local environment that may influence site occupancy include the height of the structure and the presence of internal or external lights that influence detection and disturbance, and the number of exits that make it easier for owls to respond to disturbance (Taylor, 1994). Additional information on how the configuration of highways, roads, urban cover and crops influence prey availability in adjacent grasslands would help determine what constitutes high quality foraging habitat and could also help improve the site occupancy models.

In summary, this study demonstrates barn owls have some resiliency to changes in the agricultural landscape since they continue to occupy a high proportion of suitable sites within the Fraser Valley. However, we also found that they are less likely to continue to occupy historical sites with high traffic exposure and somewhat less likely to occupy sites close to highways. This study therefore adds to the growing evidence that the urbanization of agricultural landscapes negatively impacts the associated wildlife (Fahrig & Rytwinski, 2009) and highlights the importance of managing the configuration of the landscape, rather than simply overall habitat availability.

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References

Agricultural Land Reserve. (2009). *Agricultural Land Reserve statistics*. Retrieved from http://www.alc.gov.bc.ca/alr/stats/Table 2_incl-excl_RDallyears.pdf

Andrusiak, L. A. (1994). *Nesting and roosting habitat and breeding biology of the barn owl (Tyto alba) in the Lower Mainland of British Columbia* (M.Sc. thesis). University of British Columbia.

Aschwanden, J., Holzgang, O., & Jenni, L. (2007). Importance of ecological compensation areas for small mammals in intensively farmed areas. *Wildlife Biology*, *13*, 150–158.

Barber, J. R., Crooks, K. R., & Fristrup, K. M. (2010). The costs of chronic noise exposure for terrestrial organisms. *Trends in Ecology and Evolution*, *25*, 180–189.

Baudvin, H. (1997). Barn owl (*Tyto alba*) and long-eared owl (*Asio otus*) mortality along motorways in Bourgogne-Champagne: Report and suggestions. In J. R. Duncan, D. H. Johnson, & T. H. Nicholls (Eds.), *Biology and conservation of owls of the Northern Hemisphere: 2nd international symposium, Winnipeg, Manitoba*. St. Paul, MN: U.S. Department of Agriculture, Forest Service.

Benton, T. G., Vickery, J. A., & Wilson, J. D. (2003). Farmland biodiversity: Is habitat heterogeneity the key? *Trends in Ecology and Evolution*, *18*, 182–188.

Bignal, E. M., & McCracken, D. I. (1996). Low-intensity farming systems in the conservation of the countryside. *Journal of Applied Ecology*, *33*, 413–424.

Bohlen, P. J., & House, G. (2009). *Sustainable agroecosystem management: Integrating ecology, economics, and society*. Boca Raton: CRC Press.

Bolger, D. T., Scott, T. A., & Rotenberry, J. T. (2001). Use of corridor-like landscape structures by bird and small mammal species. *Biological Conservation*, *102*, 213–224.

Bond, G., Burnside, N. G., Metcalfe, D. J., Scott, D. W., & Blamire, J. (2005). The effects of land-use and landscape on barn owl (*Tyto alba*) breeding success in southern England, U.K. *Landscape Ecology*, *20*, 555–566.

Boves, T. J. (2007). *The effects of roadway mortality on barn owls in southern Idaho and a study of ornamentation in North American barn owls* (M.Sc. thesis). Boise State University.

Brennan, L. R., & Kuvlesky, W. P., Jr. (2005). North American grassland birds: An unfolding conservation crisis? *Journal of Wildlife Management*, *69*, 1–13.

British Columbia Ministry of Transportation. (2008). *Traffic data program*. Retrieved from http://www.gov.bc.ca/trafficData/index.asp

Buckley, J., & Goldsmith, J. G. (1975). The prey of the barn owl (*Tyto alba alba*) in East Norfolk. *Mammal Review*, *5*, 13–16.

Bunn, D. S., Warburton, A. B., & Wilson, R. D. S. (1982). *The barn owl*. Vermillion: Buteo Books.

Burnham, K. P., & Anderson, D. R. (2002). *Model selection and inference: A practical information-theoretic approach*. New York: Springer.

Butet, A., & Leroux, A. B. A. (2001). Effects of agricultural development on vole dynamics and conservation of Montagu's harrier in western French wetlands. *Biological Conservation*, *100*, 289–295.

Campbell, R. W., Dawe, N. K., McTaggart-Cowan, I., Cooper, J. M., Kaiser, G. W., & McNall, M. C. E. (1990). *The birds of British Columbia: Diurnal birds of prey through woodpeckers*. Vancouver: UBC Press.

Chamberlain, D. E., & Crick, H. Q. P. (2003). Temporal and spatial associations in aspects of reproductive performance of Lapwings *Vanellus vanellus* in the United Kingdom, 1962–99. *Ardea*, *91*, 183–196.

Colvin, B. A. (1984). *Barn owl foraging behavior and secondary poisoning hazard from rodenticide use on farms* (Ph.D. thesis). Bowling Green University.

Colvin, B. A. (1985). Common barn-owl population decline in Ohio and the relationship to agricultural trends. *Journal of Field Ornithology*, *56*, 224–235.

COSEWIC. (2010). *COSEWIC assessment and status report on the barn owl Tyto alba (Eastern population and Western population) in Canada*. Ottawa, Canada: Committee on the Status of Endangered Wildlife in Canada, Ottawa, Canada. Retrieved from http://www.sararegistry.gc.ca/document/dspDocument_e.cfm?documentID=2260

Derting, T. L., & Cranford, J. A. (1989). Physical and behavioral correlates of prey vulnerability to barn owl (*Tyto alba*) predation. *American Midland Naturalist*, *121*, 11–20.

Donald, P. F., Sanderson, F. J., Burfield, I. J., & van Bommel, F. P. J. (2006). Further evidence of continent-wide impacts of agricultural intensification on European farmland birds, 1990–2000. *Agricultural Ecosystems and Environments*, *116*, 189–196.

Environment Canada. (2010). *North American breeding bird survey – Canadian results and analysis*. Gatineau, Quebec: Environment Canada. Retrieved from http://www.ec.gc.ca/reom-mbs/default.asp?lang=En&n=0D74F35F-1

Fahrig, L., & Rytwinski, T. (2009). Effects of roads on animal abundance: An empirical review and synthesis. *Ecology and Society*, *14*, 21.

Fajardo, I. (2001). Monitoring non-natural mortality in the barn owl (*Tyto alba*) as an indicator of land use and social awareness in Spain. *Biological Conservation*, *97*, 143–149.

Fast, S. J., & Ambrose, H. W. (1976). Prey preference and hunting habitat selection in the barn owl. *American Midland Naturalist*, *96*, 503–507.

Filippi-Codaccioni, O., Devictor, V., Clobert, J., & Julliard, R. (2008). Effects of age and intensity of urbanization on farmland bird communities. *Biological Conservation*, *141*, 2698–2707.

Forman, R. T. T., Sperling, D., Bissonette, J. A., Clevenger, A. P., Cutshall, C. D., Dale, V. H., et al. (2003). *Road ecology: Science and solutions*. Washington: Island Press.

Fortin, M.-J., & Dale, M. R. T. (2005). *Spatial analysis: A guide for ecologists*. Cambridge: Cambridge University Press.

Francis, C. D., Ortega, C. P., & Cruz, A. (2009). Noise pollution changes avian communities and species interactions. *Current Biology*, *19*, 1415–1419.

Fuller, R. J., Gregory, R. D., Gibbons, D. W., Marchant, J. H., Wilson, J. D., Bailie, S. R., et al. (1995). Population declines and range contractions among lowland farmland birds in Britain. *Conservation Biology*, *9*, 1425–1441.

GeoBC. (2010). *Digital road atlas 2010*. Retrieved from http://geobc.gov.bc.ca

Gorman, M. L., & Reynolds, P. (1993). The impact of land-use change on voles and raptors. *Mammal Review*, *23*, 121–126.

Hindmarch, S. (2010). *The effects of landscape composition and configuration on barn owl (Tyto alba) distribution, diet and productivity in the Fraser Valley, British Columbia* (M.Sc. thesis). Simon Fraser University.

Jaeger, J. A. G., Bowman, J., Brennan, J., Fahrig, L., Bert, D., Bouchard, J., et al. (2005). Predicting when animal populations are at risk from roads: An interactive model of road avoidance behavior. *Ecological Modeling*, *185*, 329–348.

Lodé, T. (2000). Effect of a motorway on mortality and isolation of wildlife populations. *Ambrio*, *29*, 163–166.

Marti, C. D., Poole, A. F., & Bevier, L. R. (2005). Barn owl (*Tyto alba*). In A. Poole (Ed.), *The birds of North America online*. Ithaca: Cornell Lab of Ornithology. http://dx.doi.org/10.2173/bna.1. Retrieved from http://bna.birds.cornell.edu.proxy.lib.sfu.ca/bna/species/001

Marti, C. D., Wagner, P. W., & Denne, K. W. (1979). Nest boxes for the management of barn owls. *Wildlife Society Bulletin*, *7*, 145–148.

McGregor, R. L., Bender, D. J., & Fahrig, L. (2008). Do small mammals avoid roads because of the traffic? *Journal of Applied Ecology*, *45*, 117–123.

Menard, S. (1995). *Applied logistic regression analysis*. Thousand Oaks: Sage.

Metro Vancouver. (2008). *Metro Vancouver*. Retrieved from http://www.metrovancouver.org/

Nagelkerke, N. J. D. (1991). A note on a general definition of the coefficient of determination. *Biometrika*, *78*, 691–692.

Newton, I. (2004). The recent declines of farmland bird populations in Britain: An appraisal of causal factors and conservation actions. *Ibis*, *146*, 579–600.

Newton, I., Wyllie, I., & Asher, A. (1991). Mortality causes in British barn owls (*Tyto alba*) with a discussion of aldrin–dieldrin poisoning. *Ibis*, *133*(2), 162–169.

North, M. E., & Teversham, J. M. (1983). The vegetation of the flood plains of the lower Fraser, Serpentine and Nicomekl rivers, 1859 to 1890. *Syesis*, *17*, 47–66.

Percival, S. (1991). Populations trends in British barn owls. A review of some possible causes. *British Wildlife*, *2*, 131–140.

Peterjohn, B. G. (2003). Agricultural landscape: Can they support healthy bird populations as well as farm products? *Auk*, *120*, 14–19.

Preston, M. I., & Powers, G. A. (2006). High incidence of vehicle-induced owl mortality in the Lower Mainland and Central Fraser Valley, British Columbia. *Wildlife Afield*, *3*, 15–23.

Ramsden, D. J. (1998). Effects of barn conversions on local populations of barn owl (*Tyto alba*). *Bird Study*, *45*, 68–76.

Ramsden, D. J. (2003). *Barn owls and major roads. Results and recommendations from a 15-year research project*. Ashburton: Barn Owl Trust.

Reijnen, R., Foppen, R., ter Braak, C., & Thissen, J. (1995). The effects of car traffic on breeding bird populations in Woodland. III. Reduction of density in relation to the proximity of main roads. *Journal of Applied Ecology*, *32*, 187–202.

Robinson, R. A., & Sutherland, W. J. (2002). Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, *39*, 157–176.

Salvati, L., Ranazzi, L., & Manganaro, A. (2002). Habitat preferences, breeding success, and diet of the barn owl (*Tyto alba*) in Rome: Urban versus rural territories. *Journal of Raptor Research*, *36*, 224–228.

Shawyer, C. R. (1998). *The barn owl*. Chelmsford: Arlequin Press.

Shawyer, C. R., & Shawyer, V. M. (1995). *An investigation of the barn owl population within the Arun and Western Rother catchments*. London: Hawk and Owl Trust.

Siemers, B. M., & Schaub, A. (2011). Hunting at the highway: Traffic noise reduces foraging efficiency in acoustic predators. *Proceedings of the Royal Society B*, *278*, 1646–1652.

Siriwardena, G. M., Robinson, R. A., & Crick, H. Q. P. (2002). Status and population trends of the house sparrow *Passer domesticus* in Great Britain. In H. Q. P. Crick, R. A. Robinson, G. F. Appleton, N. A. Clark, & A. D. Rickard (Eds.), *Investigation into the causes of the decline of starlings and house sparrows in Great Britain* (pp. 33–52). Thetford: British Trust for Ornithology.

Statistics Canada. (2006). *Census of agriculture*. Retrieved from http://www.statcan.gc.ca/ca-ra2006/

Tattersall, F. H., Avundo, A. V., Manley, A. J., Hart, B. J., & MacDonald, D. W. (2000). Managing set asides for field voles (*Microtus agrestis*). *Biological Conservation*, *96*, 123–128.

Taylor, I. R. (1994). *Barn owls: Predator–prey relationships and conservation*. Cambridge: Cambridge University Press.

Toms, M. P., Crick, H. Q. P., & Shawyer, C. R. (2001). The status of breeding barn owls *Tyto alba* in the United Kingdom 1995–1997. *Bird Study*, *48*, 23–37.

Underhill, J. E., & Angold, P. G. (2000). Effects of road on wildlife in an intensively modified landscape. *Environmental Review*, *8*, 21–39.

Webster, J. A. (1973). Seasonal variation in mammal contents of barn owl castings. *Bird Study*, *20*, 185–196.

Wilson, J. D., Boyle, J., Jackson, D. B., Lowe, B., & Wilkinson, N. I. (2007). Effect of cereal harvesting method on a recent population decline of Corn Buntings *Emberiza calandra* on the Western Isles of Scotland. *Bird Study*, *54*, 362–370.