



Review

Analysis of trends and agricultural drivers of farmland bird declines in North America: A review



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ABSTRACT

Globally, agriculture has intensified during the past 50 years due to increased mechanization, changes in the timing of farming operations, grassland conversion to cropland, and increased agrochemical inputs. Birds associated with farmlands and grasslands in North America have experienced severe declines over the last several decades, prompting the need for a comprehensive review of the drivers, mechanisms and magnitude of effects on bird populations. Here we evaluated changes in North American farmland bird populations over time and conducted a systematic review and analysis of the published literature to identify the major causes. Based on North American Breeding Bird Survey data, populations of 57 of 77 (74%) farmland-associated species decreased from 1966 to 2013. Multiple species exhibited highly congruent declines during the 1960s–1980s – a period with rapid changes in farming practices to low tillage systems, heavy pesticide use and widespread conversion of grassland habitat to cropland. The most severe declines occurred in aerial insectivorous birds (average change of –39.5% from 1966 to 2013), followed by grassland (–20.8%) and shrubland (–16.5%) bird species. Direct agricultural drivers impacting bird abundance, survival, and reproduction include loss of natural habitats, interference from farming equipment, and direct mortality or sublethal effects from pesticide exposure. Subtle interference with behaviour or physiology are reported through indirect drivers such as reduced food supplies, sublethal pesticide toxicity, habitat fragmentation and alteration, and disturbance. Indirect effects are likely significant for many species, particularly aerial insectivores, but detailed mechanistic studies are lacking. Our review of 122 studies found that pesticides (42% of all studies), followed by habitat loss or alterations (27%), were most predominant in negatively affecting farmland birds, with pesticides (93% negative) and mowing/harvesting (81% negative) having the most consistently negative effects. Modifications to farmland management such as reducing pesticide inputs through integrated pest management and maintaining or restoring uncultivated field margins and native habitat could positively influence farmland birds without significantly reducing agricultural crop yields.

1. Introduction: agricultural intensification and birds as indicators of change

Agriculture has been identified as the largest global extinction threat to birds (Green et al., 2005). Farmland management and mechanization worldwide have dramatically changed since the 1960s, affecting many wildlife species that depend on farmland habitat. While the area in agricultural land use has been either stable or decreasing over the past 50 years (Table 1), there has been a shift from diverse mixed-farming systems to larger farms that specialize in either livestock or crop-based production with an increase in chemical inputs and overall reduction in landscape heterogeneity (Benton et al., 2003).

Notably, there has been the steady decline in the number of individual farms but an increase in farm size – Canadian farms almost doubled in size (+49% area) and US farms are 16% larger now than in the 1960s (Table 1). Continuous cropping has become widespread since the 1970s with increased reliance on agrochemicals (Fuller et al., 1995). Extensive use of conservation tillage, where crop stubble from the previous year is left untilled to reduce soil erosion, frequently results in higher herbicide use for weed control (Rodgers and Wooley, 1983). Other changes include loss of natural habitats (i.e. field margins and wetlands), increased mechanization, and changes in the timing of farming activities, with a marked reduction in spring sowing of cereals, as well as earlier planting and harvesting (O'Connor and Shrubbs, 1986; Warner, 1994;

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Table 1

Statistics showing decadal changes in agriculture in Canada, 1966–2016, and the United States, 1964–2007.

(source: Canadian Census of Agriculture [Statistics Canada, 2006; Statistics Canada, 2016], USDA Census of Agriculture [US Department of Agriculture USDA, 1982; USDA, 2012]).

Canada	1966	1976	1986	1996	2006	2016
Total farm area (ha)	70,464,083	68,660,645	67,825,757	68,054,956	67,586,741	64,232,948
Total number of farms	430,503	338,552	293,089	276,548	229,373	193,492
Average farm size (ha)	163	202	231	246	295	315
% of farm area in crops	39.7	41.3	48.9	51.3	53.1	60.0
United States	1964	1974	1987	1997	2007	2017
Total farm area (ha)	449,268,645	411,577,583	390,307,414	386,374,629	373,158,947	NA
Total number of farms	3,157,857	2,314,013	2,087,759	2,215,876	2,204,792	NA
Average farm size (ha)	142	178	187	174	169	NA
% of farm area in crops	39.1	43.3	46.0	46.6	44.1	NA

Stoate, 1996; Chamberlain et al., 2000). These changes in farming practices are collectively referred to as agricultural intensification (Donald et al., 2001).

Globally, agricultural intensification appears to coincide with periods of major avian declines – a linkage that has been well established in European countries (see Vickery et al., 2001; Newton, 2004). From the 1960s to the 1990s, reductions in distribution and abundance of farmland bird species in Britain were greater than those associated with any other habitat, with 86% (24 of 28 species) exhibiting reduced distributions and 83% (15 of 18 species) experiencing declines in abundance; population declines of greater than 50% were estimated for seven species (Fuller et al., 1995). Similar trends are apparent for North American farmland bird populations (Murphy, 2003; Brennan and Kuvlesky, 2005; Stephens, 2016). Common farmland birds such as vesper sparrows (*Poocetes gramineus*), eastern meadowlarks (*Sturnella neglecta*), and bobolinks (*Dolichonyx oryzivorus*) experienced population declines of 77%, 73%, and 71%, respectively, in some regions of Canada from 1968 to 2006 (Kirk et al., 2011). The temporal consistency between avian declines and agricultural intensification could signal a causal link.

To our knowledge, relationships between agriculture and avian declines have not been comprehensively reviewed for North America, with few exceptions (see Rodenhouse et al., 1995; Lacher et al., 2010). Therefore, our goal was to review and evaluate observational and experimental support for linkages and potential mechanisms involved in agriculturally-driven avian declines in North America.

2. Methods

2.1. Population trends of farmland birds

We reviewed available data on the population trends and agricultural effects for 77 North American breeding bird species inhabiting farmlands. The list of “grassland birds” was defined by the North American Breeding Bird Survey (BBS) ($n = 28$) and expanded to include species reported as farmland-dependent (Murphy, 2003) ($n = 60$, with some overlap with BBS species). Each species was categorized by foraging habitat into aerial ($n = 6$), grassland ($n = 42$) or shrubland ($n = 29$) based on Murphy (2003). Species with obligate dependence on aquatic habitats for nesting or foraging, such as Anseriformes, or closed forests, were excluded. Population changes were determined for each species using published annual BBS trend data from 1966 to 2013 (Sauer et al., 2014; Table 2). To construct figures, we determined the percent change per year based on the average annual trend for each decade (i.e. 1966–1975, 1976–1985, etc.; e.g. all years from 1966 to 1975 were assigned the average decadal trend). We then used these estimates to project population trends by habitat group (aerial, grassland or shrubland species) as well as convergence and divergence of individual species trends in response to historical changes in agriculture by decade.

2.2. Agricultural drivers of farmland bird declines

We hypothesized all potential drivers of agriculturally-mediated farmland declines with direct and indirect effects on abundance, reproduction, and survival and classified these as follows: 1) habitat loss and fragmentation through land conversion, 2) mowing and harvesting operations, 3) livestock grazing disturbance, 4) pesticide lethal and sublethal toxicity, and 5) changes in food supply (Fig. 1). While previous reviews on farmland birds in Europe (e.g. Tucker and Evans, 1997) contained information on all potential threats, our review focuses on those related to agriculture. This review excludes deliberate lethal killing practices to reduce crop damage, though these actions may have contributed to declines in bird species perceived as crop pests (e.g. rusty blackbirds [*Euphagus carolinus*, Greenberg et al., 2011], dickcissels [*Spiza americana*, Basili and Temple, 1999], bobolinks [Renfrew and Saavedra, 2007]).

Direct and indirect threats of agricultural intensification are present on breeding and wintering grounds, but we emphasize impacts on the breeding grounds in Canada and the US. For each threat, there are direct effects of agricultural intensification on reproduction and survival of birds inhabiting farmland habitats, in addition to complex indirect effects. Direct effects are those that result in mortality or destruction of eggs, young, or adults, triggering reductions in reproductive success and survival. Indirect effects also result in mortality or destruction of eggs, young, or adults, but occur through an intermediary action(s) (Fig. 1).

To determine the extent of agricultural impacts on avian species, a literature search was completed using all years in Web of Science™ (accessed September 2013 to February 2017). Searches were limited to the US and Canada as data for Mexico were scarce, and search terms included “birds”, “agriculture”, and [effect], where [effect] was either conservation, fertilizer, field margins, food supply, foraging, grazing, habitat fragmentation, habitat loss, herbicides, insecticides, management, mowing, pesticides, physiology, or predation, as well as [species] and “agriculture” or “agricultural”, with [species] including the 77 species listed in Table 2. A summary of available published data for the 77 species organized by driver is available in Appendix A. Factors for inclusion in the quantitative analysis included studies that met the following criteria: 1) results supported by p-values for an identified species or set of species, 2) reported one or more of the “effect parameters” (below), and 3) evaluated one or more of the drivers (habitat loss and fragmentation, pesticide toxicity, mowing and harvesting, etc.). These data were individually reviewed to identify the number of unique species and studies for each driver, whether the effect was positive (significant positive response), neutral (no effect) or negative (significant negative response) on the following “effect parameters” including survival, reproduction, abundance, density, occurrence, behaviour, physiology, and predation risk. For presentation purposes, we grouped those studies reporting abundance, density and occurrence into a single category.

3. Declining trends of North American farmland birds

Farmland birds are experiencing faster population declines than birds associated with any other biome based on analyses of North American Breeding Bird Survey (BBS) data (Murphy, 2003; Sauer et al., 2014). Here, we found that 57 of 77 (74%) species associated with farmland habitats decreased from 1966 to 2013; declines were most severe among farmland aerial insectivores (average change of -39.5% from 1966 to 2013), followed by grassland (-20.8%) and shrubland (-16.5%) bird species (Fig. 2). Aerial insectivores as a guild are declining most rapidly for reasons that are poorly understood, but may be related to modified flying insect prey through increased cropping intensity, loss of field margins, increased use and toxicity of insecticides and drainage or degradation of surface waters (Nebel et al., 2010;

Hallmann et al., 2014). In the United Kingdom, changes in agricultural practices were associated with loss of lepidopteran moths and butterflies (Thomas et al., 2004) and the same mechanism has been proposed for aerial insectivores (Benton et al., 2002). Overall, avian declines were comparable in both Canada and the US (Table 2), although clear species- and region-specific differences highlight the need for further detailed analyses (Michel et al., 2016). The persistent decline of farmland birds is rare in avian species of other habitats, suggesting agricultural-driven effects as probable causes (Donald et al., 2006).

Given there are 67 million hectares (7.3% of total land area but regionally concentrated) of agricultural land in Canada, and 379 million hectares (41.5% of total land area) in the US (Statistics Canada, 2011; USDA, 2014), understanding potential links between agriculture and avian declines may yield new insights into ways of mitigating

Table 2

Summary of average yearly population trends (with 2.5 and 97.5% credible intervals (CI) for 77 farmland-associated bird species in Canada and the United States based on North American Breeding Bird Survey estimates, 1966–2013. Species are grouped (guilds) by aerial insectivores, and grassland and shrubland birds. Within guilds, species are ranked in order of survey-wide trend estimates from positive to negative.

Species	Average trend per year ^a		
	Canada (CI) ^b	United States (CI)	Survey-wide (CI)
Aerial			
Cliff swallow (<i>Petrochelidon pyrrhonota</i>)	-4.66 (-8.83, -2.82)	1.20 (0.54, 1.63)	0.25 (-2.24, 0.88)
Purple martin (<i>Progne subis</i>)	-1.15 (-2.56, 0.22)	-0.86 (-3.27, -0.43)	-0.87 (-3.24, -0.45)
Barn swallow (<i>Hirundo rustica</i>)	-3.67 (-4.06, -3.33)	-0.51 (-0.64, -0.38)	-1.27 (-1.44, -1.12)
Tree swallow (<i>Tachycineta bicolor</i>)	-2.23 (-2.85, -1.75)	0.09 (-0.32, 0.42)	-1.37 (-1.91, -1.03)
Common nighthawk (<i>Chordeiles minor</i>)	-4.05 (-10.01, -2.47)	-1.88 (-2.61, -1.54)	-2.06 (-7.16, -1.69)
Bank swallow (<i>Riparia riparia</i>)	-8.05 (-9.67, -6.13)	-2.30 (-4.17, -0.80)	-5.83 (-7.37, -4.38)
Guild average	-3.97	-0.71	-1.86
Grassland			
Greater prairie-chicken (<i>Tympanuchus cupido</i>)	NA	4.13 (-1.19, 8.40)	4.13 (-1.19, 8.40)
Turkey vulture (<i>Cathartes aura</i>)	7.80 (3.61, 9.83)	2.33 (1.99, 2.69)	2.38 (2.05, 2.74)
Eastern bluebird (<i>Sialia sialis</i>)	2.02 (0.19, 3.37)	1.92 (1.69, 2.13)	1.92 (1.69, 2.13)
Barn owl (<i>Tyto alba</i>)	NA	1.62 (-1.71, 3.96)	1.62 (-1.71, 3.96)
Carolina wren (<i>Thryothorus ludovicianus</i>)	4.87 (-0.66, 10.79)	1.35 (1.15, 1.53)	1.35 (1.15, 1.53)
Sedge wren (<i>Cistothorus platensis</i>)	0.86 (-2.13, 2.68)	0.85 (-0.14, 1.67)	0.81 (-0.94, 1.71)
Ferruginous hawk (<i>Buteo regalis</i>)	0.08 (-2.35, 2.26)	0.92 (0.06, 1.72)	0.73 (-0.20, 1.52)
Upland sandpiper (<i>Bartramia longicauda</i>)	-0.29 (-2.08, 0.94)	0.60 (0.10, 1.10)	0.49 (-0.02, 0.96)
Wilson's snipe (<i>Gallinago delicata</i>)	0.52 (-0.17, 1.17)	-0.18 (-0.64, 0.32)	0.34 (-0.19, 0.83)
Long-billed curlew (<i>Numenius americanus</i>)	-0.56 (-1.90, 0.85)	0.65 (-1.14, 1.62)	0.34 (-1.17, 1.20)
Sharp-tailed grouse (<i>Tympanuchus phasianellus</i>)	-0.73 (-3.04, 1.17)	0.76 (-1.55, 2.72)	0.16 (-1.79, 1.67)
House wren (<i>Troglodytes aedon</i>)	-0.09 (-0.52, 0.32)	-0.15 (-0.03, 0.31)	0.07 (-0.12, 0.24)
Mourning dove (<i>Zenaidura macroura</i>)	0.65 (0.22, 1.09)	-0.42 (-0.55, -0.29)	-0.38 (-0.50, -0.26)
Dickcissel (<i>Spiza americana</i>)	20.2 (-9.29, 75.75)	-0.62 (-1.13, -0.20)	-0.62 (-1.13, -0.20)
Brown-headed cowbird (<i>Molothrus ater</i>)	-1.59 (-2.51, -1.11)	-0.50 (-0.67, -0.35)	-0.74 (-1.09, -0.58)
Vesper sparrow (<i>Pooecetes gramineus</i>)	-0.49 (-1.02, -0.01)	-1.21 (-1.60, -0.90)	-0.89 (-1.24, -0.61)
Lark sparrow (<i>Chondestes grammacus</i>)	1.63 (-0.54, 3.65)	-0.94 (-1.35, -0.51)	-0.94 (-1.34, -0.50)
Red-winged blackbird (<i>Agelaius phoeniceus</i>)	-0.91 (-1.29, -0.54)	-0.99 (-1.12, -0.86)	-0.98 (-1.09, -0.85)
Common yellowthroat (<i>Geothlypis trichas</i>)	-1.10 (-1.65, -0.61)	-0.96 (-1.06, -0.86)	-1.02 (-1.27, -0.81)
Burrowing owl (<i>Athene cunicularia</i>)	-6.00 (-10.60, -1.33)	-1.01 (-2.15, -0.14)	-1.08 (-2.19, -0.22)
Eastern kingbird (<i>Tyrannus tyrannus</i>)	-1.81 (-2.30, -1.37)	-0.96 (-1.14, -0.79)	-1.13 (-1.32, -0.96)
Henslow's sparrow (<i>Ammodramus henslowii</i>)*	-3.63 (-10.14, 2.78)	-1.13 (-2.38, -0.02)	-1.14 (-2.38, -0.02)
Killdeer (<i>Charadrius vociferous</i>)	-3.18 (-4.33, -2.70)	-0.56 (-0.74, -0.37)	-1.20 (-1.72, -0.99)
Northern harrier (<i>Circus cyaneus</i>)	-2.19 (-3.06, -1.49)	-0.59 (-1.11, -0.13)	-1.21 (-1.74, -0.81)
Savannah sparrow (<i>Passerculus sandwichensis</i>)	-1.28 (-1.65, -0.90)	-1.25 (-1.66, -0.90)	-1.27 (-1.58, -0.97)
Western meadowlark (<i>Sturnella neglecta</i>)	-2.13 (-2.63, -1.65)	-1.20 (-1.51, -0.92)	-1.30 (-1.59, -1.05)
American kestrel (<i>Falco sparverius</i>)	-2.71 (-4.43, -1.58)	-1.10 (-1.37, -0.86)	-1.65 (-2.53, -1.22)
Cassin's sparrow (<i>Peucaea cassinii</i>)	NA	-1.97 (-3.60, -0.99)	-1.97 (-3.60, -0.99)
Bobolink (<i>Dolichonyx oryzivorus</i>)	-3.65 (-5.07, -3.18)	-1.00 (-1.38, -0.59)	-2.04 (-3.07, -1.70)
Le Conte's sparrow (<i>Ammodramus leconteii</i>)	-2.27 (-3.73, -0.69)	1.33 (-0.84, 3.42)	-2.18 (-3.62, -0.71)
Horned lark (<i>Eremophila alpestris</i>)	-4.81 (-5.77, -3.88)	-1.76 (-2.11, -1.23)	-2.38 (-2.82, -1.92)
Grasshopper sparrow (<i>Ammodramus savannarum</i>)	-1.21 (-8.34, 1.83)	-2.82 (-3.42, -2.33)	-2.83 (-3.76, -2.32)
Baird's sparrow (<i>Ammodramus bairdii</i>)	-2.65 (-4.76, -0.58)	-3.51 (-5.22, -1.09)	-2.93 (-4.52, -1.31)
Mountain plover (<i>Charadrius montanus</i>)*	NA	-3.11 (-7.82, -0.80)	-3.11 (-7.82, -0.80)
Short-eared owl (<i>Asio flammeus</i>)	-4.98 (-10.78, -1.17)	-2.21 (-5.31, -0.10)	-3.26 (-7.71, -1.11)
Eastern meadowlark (<i>Sturnella magna</i>)	-3.21 (-3.67, -2.77)	-3.35 (-3.73, -3.08)	-3.34 (-3.71, -3.08)
Loggerhead shrike (<i>Lanius ludovicianus</i>)	-2.88 (-4.22, -1.52)	-3.21 (-3.50, -2.91)	-3.49 (-3.49, -2.91)
Sprague's pipit (<i>Anthus spragueii</i> **)	-3.87 (-5.24, -2.64)	-1.05 (-3.24, 1.15)	-3.51 (-4.83, -2.34)
Northern bobwhite (<i>Colinus virginianus</i>)*	-21.4 (-25.97, -17.12)	-4.08 (-4.38, -3.84)	-4.08 (-4.38, -3.84)
Lark bunting (<i>Calamospiza melanocorys</i>)	-6.41 (-10.62, -1.71)	-4.03 (-6.12, -2.38)	-4.10 (-6.12, -2.38)
Chestnut-collared longspur (<i>Calcarius ornatus</i>)*	-5.49 (-7.04, -3.73)	-3.65 (-4.57, -2.66)	-4.35 (-5.30, -3.33)
McCown's longspur (<i>Rhynchophanes mccownii</i>)	-8.24 (-11.67, -3.76)	-0.99 (-3.41, 1.47)	-6.20 (-8.90, -2.85)
Guild average	-1.60	-0.83	-1.21

(continued on next page)

Table 2 (continued)

		Average trend per year ^a		
Species		Canada (CI) ^b	United States (CI)	Survey-wide (CI)
Shrub	Red-tailed hawk (<i>Buteo jamaicensis</i>)	0.51 (−0.07, 1.04)	1.78 (1.59, 1.96)	1.46 (1.23, 1.66)
	Blue grosbeak (<i>Guiraca caerulea</i>)	NA	0.81 (0.53, 1.02)	0.81 (0.53, 1.02)
	White-eyed vireo (<i>Vireo griseus</i>)	NA	0.51 (0.30, 0.71)	0.51 (0.30, 0.71)
	Eastern phoebe (<i>Sayornis phoebe</i>)	−1.46 (−2.90, −0.27)	1.25 (1.06, 1.43)	0.50 (−0.20, 0.91)
	Northern cardinal (<i>Cardinalis cardinalis</i>)	3.89 (3.15, 4.69)	0.33 (0.24, 0.42)	0.34 (0.25, 0.43)
	American crow (<i>Corvus brachyrhynchos</i>)	−0.17 (−0.51, 0.15)	0.35 (0.24, 0.45)	0.16 (0.00, 0.29)
	American robin (<i>Turdus migratorius</i>)	−0.10 (−0.30, 0.09)	0.32 (0.23, 0.40)	0.14 (0.04, 0.24)
	Cedar waxwing (<i>Bombycilla cedrorum</i>)	−0.52 (−1.62, 0.24)	0.69 (0.28, 1.02)	−0.04 (−0.90, 0.46)
	Gray catbird (<i>Dumetella carolinensis</i>)	−0.41 (−0.77, −0.06)	−0.04 (0.15, 0.07)	−0.09 (−0.19, 0.02)
	American goldfinch (<i>Carduelis tristis</i>)	−0.62 (−1.04, −0.25)	0.07 (−0.11, 0.23)	−0.18 (−0.38, −0.00)
	Yellow warbler (<i>Dendroica petechia</i>)	−0.76 (−1.19, −0.38)	−0.20 (−0.39, −0.01)	−0.53 (−0.82, −0.29)
	Blue-winged warbler (<i>Vermivora pinus</i>)	6.87 (3.07, 10.92)	−0.64 (−1.28, 0.13)	−0.58 (−1.21, 0.18)
	Northern mockingbird (<i>Mimus polyglottos</i>)	0.71 (−1.69, 2.87)	−0.59 (−0.75, −0.43)	−0.59 (−0.75, −0.43)
	Chipping sparrow (<i>Spizella passerina</i>)	−1.19 (−1.68, −0.76)	0.07 (−0.11, 0.21)	−0.59 (−0.94, −0.34)
	Yellow-breasted chat (<i>Icteria virens</i>)	1.68 (−2.87, 6.03)	−0.67 (−0.87, −0.47)	−0.67 (−0.87, −0.47)
	Song sparrow (<i>Melospiza melodia</i>)	−1.06 (−1.39, −0.78)	−0.48 (−0.61, −0.37)	−0.74 (−0.90, −0.59)
	Indigo bunting (<i>Passerina cyanea</i>)	0.89 (0.09, 1.59)	−0.80 (−0.89, −0.70)	−0.78 (−0.87, −0.69)
	Orchard oriole (<i>Icterus spurius</i>)	4.76 (2.18, 6.63)	−0.85 (−1.14, −0.60)	−0.84 (−1.12, −0.59)
	Brown thrasher (<i>Toxostoma rufum</i>)	−1.13 (−1.71, −0.59)	−1.02 (−1.15, −0.89)	−1.02 (−1.16, −0.90)
	Baltimore oriole (<i>Icterus galbula</i>)	−3.20 (−4.00, −2.52)	−0.75 (−1.00, −0.55)	−1.35 (−1.70, −1.09)
	Eastern towhee (<i>Pipilo erythrophthalmus</i>)	−0.87 (−2.10, 0.32)	−1.38 (−1.52, −1.25)	−1.38 (−1.52, −1.25)
	Willow flycatcher (<i>Empidonax traillii</i>)	−1.80 (−2.51, −1.05)	−1.25 (−1.72, −0.86)	−1.46 (−1.86, −1.08)
	Chestnut-sided warbler (<i>Dendroica pensylvanica</i>)	−1.64 (−2.49, −0.92)	−0.92 (−1.16, −0.68)	−1.48 (−2.18, −0.91)
	Yellow-billed cuckoo (<i>Coccyzus americanus</i>)	−1.25 (−2.87, 0.49)	−1.75 (−1.99, −1.50)	−1.75 (−1.99, −1.50)
	Common grackle (<i>Quiscalus quiscula</i>)	−0.80 (−1.22, −0.43)	−1.85 (−2.01, −1.69)	−1.76 (−1.91, −1.60)
	Prairie warbler (<i>Dendroica discolor</i>)	NA	−1.96 (−2.30, −1.62)	−1.96 (−2.30, −1.62)
Field sparrow (<i>Spizella pusilla</i>)	−1.95 (−6.59, −1.16)	−2.35 (−38.74, −2.17)	−2.36 (−38.73, −2.17)	
Red-headed woodpecker (<i>Melanerpes erythrocephalus</i>)*	−2.50 (−4.32, −0.84)	−2.48 (−2.81, −2.15)	−2.48 (−2.81, −2.16)	
Black-billed cuckoo (<i>Coccyzus erythrophthalmus</i>)	−2.04 (−5.59, 0.12)	−3.25 (−8.03, −2.52)	−2.73 (−7.56, −1.42)	
Guild average	−0.16	−0.59	−0.74	

*Near-threatened.

**Vulnerable.

^a Trends with sufficient sample sizes and CIs that do not contain 0 (and therefore are considered significant) are indicated in bold.

^b Trends for species either not present or not surveyed in Canada are represented by “NA”.

adverse agricultural land use effects. In visualizing the population trends for all 77 farmland bird species, we found a strong convergence of negative trends during 1960–1980 – a period with rapid changes in farming practices to low-tillage systems, heavy pesticide use and widespread conversion of grassland habitat to cropland (Fig. 3). From the 1960s to 1990s, single farms increased production to support 4 times more people (1960s = 25.8 people; 1990s = 100 people supported by 1 farmer) (<https://www.agclassroom.org/gan/timeline/index.htm>). This was made possible in part by the increased reliance on agrochemicals with a peak in US pesticide use by 1980 of 508,000 kg (1.12 million lbs; USDA, 2014) and an increase in area under crop in both Canada (15.9%; Statistics Canada, 2006) and the United States (2.5%, USDA 1982) between the mid-1960s to mid-1980s (Table 1). Grassland or farmland bird specialists such as barn swallows (*Hirundo rustica*), bobolinks, or chestnut-collared longspurs (*Calcarius ornatus*)

continued to decline whereas other more generalist bird species such as house wrens (*Troglodytes aedon*), American crows (*Corvus brachyrhynchos*), and some hawks (e.g. red tailed and ferruginous hawks; *Buteo jamaicensis* and *regalis*) show stable or increasing trends. Many of the migratory granivores that typically would benefit from access to agricultural food sources have also experienced consistent declines, such as Lapland longspurs (*Calcarius lapponicus*), horned larks (*Eremophila alpestris*) and red-winged blackbirds (*Agelaius phoeniceus*) that use agricultural landscapes for staging or breeding (Galle et al., 2009). Only a few resident breeding farmland species, most notably corvids, appear to benefit from increased agricultural intensity (Gregory and Marchant, 1996; Gade, 2010). However, the increased prevalence of these predatory species in agricultural landscapes may also negatively affect other songbird species.

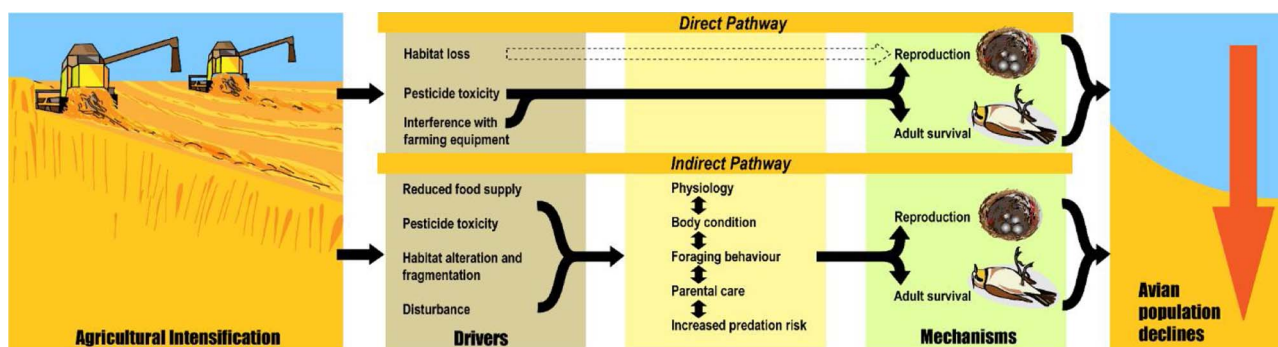


Fig. 1. Potential direct and indirect effect pathways of agricultural intensification effects on avian population declines. Reproduction includes mortality or impairment at egg and nestling/juvenile stages, whereas survival represents adult survival. Although shown separately, direct and indirect effects on reproduction and survival could act simultaneously.

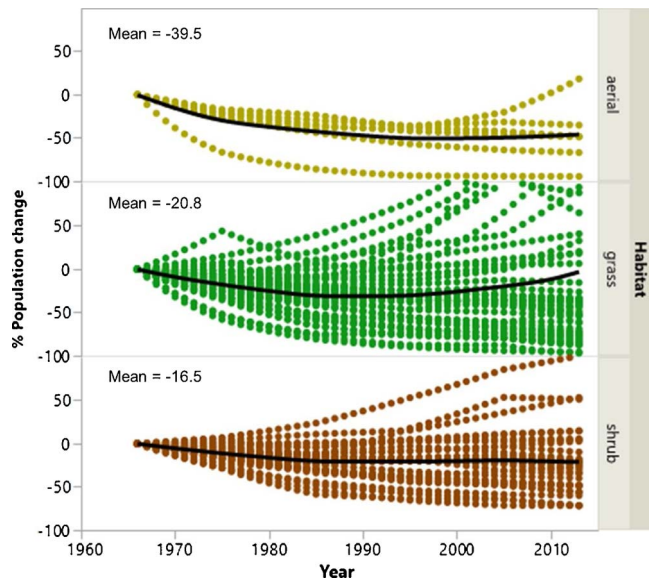


Fig. 2. Population trends of 77 North American farmland bird species based on data collected by the North American Breeding Bird Survey, 1966–2013 (Sauer et al., 2014). Dotted lines represent % change in individual species populations based on the average annual trend for each decade (1966–1975, 1976–1985, etc.). The solid black line and mean estimate represent the average trend for all species in each habitat guild over the duration of study period 1966–2013.

4. Review of potential drivers of farmland bird declines

4.1. Habitat loss and fragmentation

Habitat loss has been described as a primary driver of agriculturally-mediated grassland bird declines (Herkert et al., 1996; Vickery et al.,

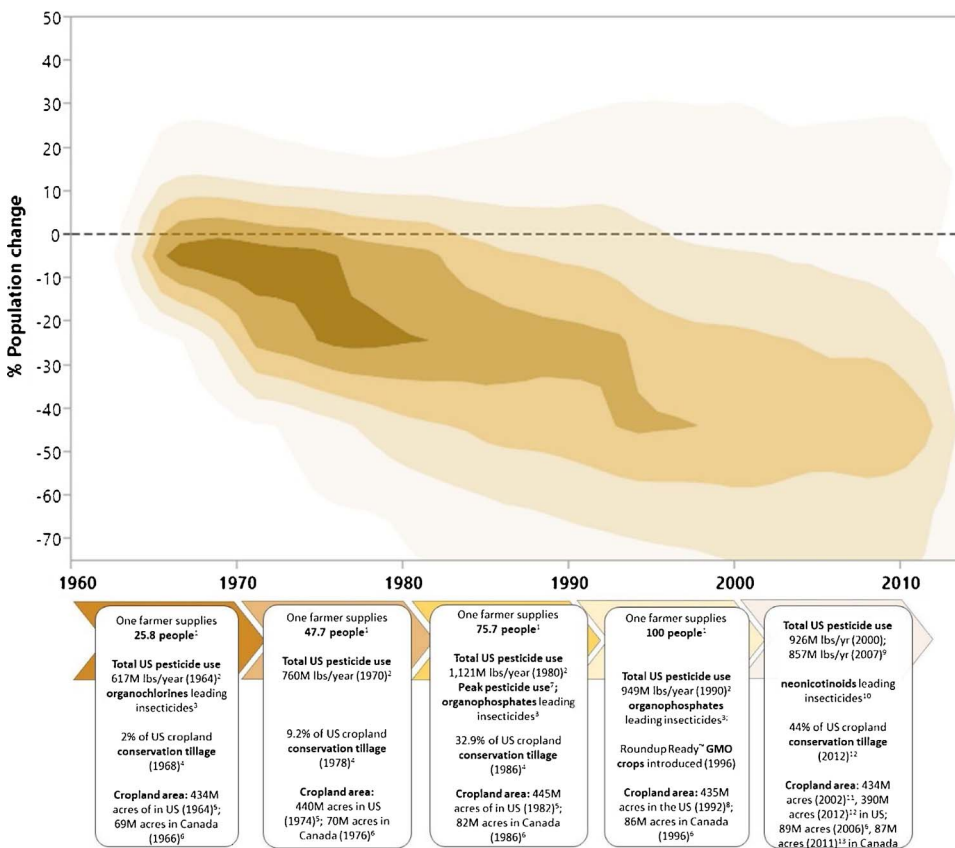


Fig. 3. Contour density plot showing population trends for 77 North American farmland bird species (grassland, shrub and aerial insectivores) from 1966 to 2013, based on the North American Breeding Bird Survey. Contours show data concentrations where trend data among species are converging (darker colours) or diverging (lighter colours). Data on bird trends are represented alongside the timing of key changes in agricultural practices through the decades (1960s–2010).

¹<https://www.agclassroom.org/gan/timeline/index.htm>, ²US EPA (2004), ³Aspelin (2003), ⁴Schertz (1988), ⁵US Department of Agriculture USDA (1982), ⁶Statistics Canada (2006), ⁷USDA (2014), ⁸USDA (1992), ⁹US EPA (2011), ¹⁰Jeschke et al. (2011), ¹¹USDA, 2002, ¹²USDA (2012), ¹³Statistics Canada (2011).

1999; Brennan and Kuvlesky, 2005). From 1982 to 1997, 97,000 km² of grasslands were lost in the US, with most converted to agriculture (Samson et al., 2004), while approximately 75% of prairie grassland in Canada had been lost due to agriculture by 1993 (Downes et al., 2011). In areas that are particularly well suited for agricultural crops such as landscapes with flat topography and rich soils, grassland loss is even more profound, with less than 0.1% of native prairie remaining (Samson and Knopf, 1994). Only 4% of tallgrass prairie remains in North America (Steinauer and Collins, 1996), while < 13% of the short-grass prairie and almost none of the tallgrass prairie remains intact in Canada (McLaughlin and Mineau, 1995). There has also been an estimated 70% loss of wetlands in the Prairie region of Canada (Canadian Wetland Inventory, 2008), with 84% of the loss attributed to agricultural drainage (Canadian Wildlife Service, 1991).

Widespread loss of grassland habitat usually produces fragmentation and isolation of remnant grasslands. Many farmland bird species are sensitive to edge effects or fragment size, so smaller, isolated grasslands are unlikely to sustain diverse bird communities (Helzer and Jelinski, 1999; Vickery et al., 1999; Kirk et al., 2011); however, effects of fragmentation on occurrence or abundance have been absent for many other species (Bayard and Elphick, 2010). Regardless, habitat fragmentation can result in higher rates of nest predation. In prairie fragments < 100 ha, 78–84% of all nests of four grassland birds (dickcissel, eastern meadowlark [*Sturnella magna*], grasshopper sparrow [*Ammodramus savannarum*], and Henslow’s sparrow [*Ammodramus henslowii*]) were depredated, compared to 54–68% of nests in fragments > 1000 ha (Herkert, 2003; also see Horn et al., 2005).

Avian species composition, abundance and diversity are influenced by compositional (i.e. number of landscape types) and configurational (i.e. complexity of the landscape) heterogeneity (Lindsay et al., 2013). Recent conversion of pastures and hayfields to row cropping in the US Midwest, along with altered cutting regimes, have resulted in unsuitable nesting habitat for several grassland species (Warner, 1994; Herkert et al., 1996). Even where pastures still exist, alfalfa monocultures have often replaced mixed

stands of grasses and legumes that had previously provided birds with native-like prairie habitat (Graber and Graber, 1963; Warner, 1994). Field margins, forest patches and hedgerows are important habitats for enhancing landscape complexity in agricultural landscapes, and providing essential refugia for diverse insect communities (Dennis and Fry, 1992; Mineau and McLaughlin, 1996). Presence of hedgerows increases abundance and diversity of beneficial insect and bird species that can provide important ecosystem services (Jobin et al., 2001; Morandin et al., 2014), and native upland and wetland habitats have higher relative abundance and diversity of avian species than conventional, minimum tillage, and organic farms (Shutler et al., 2000). Wetland loss and degradation can also indirectly influence reproduction of farmland birds. For example, the number of red-winged blackbird fledglings per nest increased with wetland size and abundance in the surrounding landscapes (Tozer et al., 2010), where higher fledging success was thought to result from greater food availability.

4.2. Pesticide use and toxicity

The latest estimates for 2011 and 2012 indicate U.S. pesticide usage totaled over 498 million kg (1.1 billion lbs) annually, accounting for 16% of the world market (US EPA, 2017). It has been estimated that direct pesticide mortality accounts for the deaths of approximately 67–72 million birds in the US each year (Pimentel et al., 1992; Pimentel, 2005). This estimate conservatively assumes 10% mortality among exposed birds, and excludes mortality in wintering or migrating birds. In Canada, pesticide use contributes to an estimated 2.7 million cases of avian mortality annually (Calvert et al., 2013). Mixtures of multiple insecticide, fungicide and herbicide products are common in the environment to control various pests and weeds. Although rarely studied, exposure and effects to birds may be greater than expected based on toxicity to single compounds. Exposure to sixteen current-use pesticides (including carbamates, organophosphates, and neonicotinoids) was associated with changes in thyroid physiology important for growth of tree swallow (*Tachycineta bicolor*) and eastern bluebird (*Sialia sialis*) nestlings (Mayne et al., 2005). Insect bolus samples collected from nestling tree swallows contained traces of 25 different compounds, and at least 30% of samples contained at least 1 pesticide (Haroune, 2015). Although many herbicides are considered less toxic to birds, linuron exposure in American goldfinches (*Carduelis tristis*) altered thyroid hormones and moult progression (Sughrue et al., 2008), and reduced mating success (Robertson et al., 1998).

Some of the most highly toxic pesticides to birds include the widely studied carbamates and organophosphates (Mineau et al., 1999, 2005; Mineau, 2004) capable of killing thousands of birds at a time (Mineau, 1993). Canada's Pest Management Regulatory Agency estimated between 109,000 and 958,000 birds were killed annually by carbofuran use on canola seeds alone (Segstro, 1998). Carbofuran was estimated to have killed 17 to 91 million songbirds annually in the US during peak use on corn in the 1980s (Mineau, 2005). A 5% carbofuran granule can kill a house sparrow (*Passer domesticus*) within minutes, while individuals that consume one granule of terbufos (organophosphate) have a 50% survival probability (Mineau et al., 2005). The amount of organophosphate insecticides used in the U.S. has declined more than 70%, from an estimated 31 million kg (70 million lbs) in 2000 to 9 million kg (20 million lbs) in 2012 (US EPA, 2017) largely due to increased restrictions and rising popularity of newer products like neonicotinoids.

Neonicotinoids (including imidacloprid, thiamethoxam, and clothianidin) are a class of systemic pesticides which now represent the most widely used insecticides worldwide (Jeschke et al., 2011), and are also of recent concern to farmland birds. Hallmann et al. (2014) reported a 3.5% average annual decline among 15 species of farmland birds in regions of the Netherlands where imidacloprid in water exceeded 20 ng/L. Applied primarily as a seed dressing, neonicotinoids comprise ~25% of all insecticide use, and are licensed in more than 120 countries (Jeschke et al., 2011). A single corn kernel treated with imidacloprid, or just a few seeds treated with clothianidin or

thiamethoxam, can potentially harm or kill a medium-sized bird such as a blue jay (*Cyanocitta cristata*), and as little as 1/10th of a neonicotinoid-treated corn kernel per day during egg-laying has been estimated to impair songbird reproduction (Mineau and Palmer, 2013). A captive study on white-crowned sparrows (*Zonotrichia leucophrys*) indicated ingestion of just 4–9 imidacloprid-treated seeds per day for 3 days can result in severe mass loss (17–25%) as well as temporary impairment of migratory orientation (Eng et al., 2017). Currently, there is a lack of studies looking at exposure and mortality in free-living birds that may consume neonicotinoid-treated seeds.

Survival and reproduction of birds can also be affected by sublethal pesticide toxicity through intermediate effects on behaviour, physiology, or condition. Sublethal exposure to pesticides such as the organophosphate acephate or chlorpyrifos can lead to altered songbird flight orientation (Vyas et al., 1995; Eng et al., 2017), while carbofuran can impair an individual's ability to thermoregulate (Friend and Franson, 1999), or trigger ataxia (loss of control of body movements), dyspnea (labored breathing), immobility, and opisthotonos (muscle spasms leading to arching of back) (Hudson et al., 1984), all of which may increase vulnerability to predation and starvation.

Parental care is also sensitive to pesticides. Nestling tree swallows exhibited significant increases in begging following an application of azinphos-methyl and carbaryl insecticides in apple orchards, while parents reduced the number of feeding trips to the nest following a second application (Bishop et al., 2000). Likewise, European starlings (*Sturnus vulgaris*) dosed with dicrotophos, had fewer feeding trips and lower chick mass gains within 12–24 h after dosing (Grue et al., 1982). Even if responses are temporary, such short-term effects could lower reproductive success, first year or adult survival.

4.3. Mowing and harvesting operations

Mechanization has increased over recent decades, and mowing, tilling, seeding, and harvesting are sources of direct avian mortality and nest destruction (Nocera et al., 2005; Shustack et al., 2010). Farm machinery used for soybean planting caused losses of 24.4% of 114 nests from a variety of grassland bird species; a 2-week delay in planting to early June would allow all destroyed nests to complete the nesting cycle, although soybean yields decrease for every day planting is delayed (VanBeek et al., 2014). Higher nest density was found in no-till corn and soybean fields than tilled fields (Basore et al., 1986), as was greater species richness and abundance (Castrale, 1985; Walk et al., 2010) as denser vegetation in these fields is hypothesized to lower predation risk (VanBeek et al., 2014).

Typically, nest destruction from tilling, seeding, or harvesting is less severe than that from mowing, as tilling and seeding occur before many species have established nests, and harvesting occurs after nestlings have already fledged. In contrast, mowing operations can occur repeatedly, and often overlap with the breeding season (Tews et al., 2013), so the frequency and timing of mowing of hayfields can influence the relative impact on birds. Since the 1950s, the median date of mowing has advanced by 14–21 days, resulting in an overlap with peak nesting for farmland birds in many regions (Martin and Gavin, 1995; Herkert, 1997), likely due to a shift from native warm-season to exotic cool-season grasses that produce higher biomass earlier in summer (Giuliano and Daves, 2002). Birds breeding in warm-season cultivars experience greater nest success (measured as proportion of successful nests per field) and fledge rates compared to those in cool-season grasses, as a result of lower nest destruction and predation rates (Giuliano and Daves, 2002). Vesper sparrow nest success was significantly higher in wet seasons compared to dry seasons due to delayed (~30 days) harvests in wet years (Perritt and Best, 1989).

Indirect effects of mowing and haying can affect reproduction and survival of farmland nesting birds (Diemer and Nocera, 2016). In grasshopper sparrows, 8.5% of nest failures resulted from direct interference with mowing equipment, whereas 67% failed due to subsequent

predation (Giuliano and Daves, 2002). Nests that survive haying are often abandoned or destroyed by predators (Perritt and Best, 1989; Bollinger et al., 1990). In regions of Vermont and New York, 100% of active bobolink nests and 99% of active savannah sparrow (*Passerculus sandwichensis*) nests failed during early haying operations (Perlut et al., 2006), mainly due to higher nest predation. Leaving uncut patches for nesting birds, or delaying the timing of haying, may be a promising conservation strategy to reduce the effect of harvesting disturbance on farmland birds.

4.4. Grazing disturbance

Livestock grazing can result in increased predation risk, degradation of nesting habitat, or trampling of nests by cattle (Bleho et al., 2014), as well as rare instances of apparent predation by cattle (Nack and Ribic, 2005), but is highly dependent on species and region. Grazing can also benefit some species of grassland songbirds by influencing vegetation structure to create more suitable nesting habitat for short grass specialists (Fondell and Ball, 2004). Moderate levels of grazing provided suitable habitat for Henslow's and grasshopper sparrows in New York (Smith, 1997), while in Illinois, light late-season grazing resulted in lower density of Henslow's sparrow, but higher density of dickcissels, eastern meadowlarks, and grasshopper sparrows (Walk and Warner, 2000). The same variation in response to grazing exists for nest success. While some studies have found reduced nest success as a result of grazing (Shochat et al., 2005; Sutter and Ritchison, 2005), others found no difference between grazed and ungrazed areas (Zimmerman, 1996; Klute et al., 1997). A review of nest trampling rates by Bleho et al. (2014) reported that only 1.5% of 9132 nests across 9 ecoregions were directly destroyed by cattle, which was correlated with stocking rates. As rates of nest destruction are very low, it is unlikely to be of primary importance for management actions.

4.5. Food supply and diet composition

The connection between agriculture and population effects due to changes in food availability is difficult to demonstrate. Although rarely included in ecological risk assessments, indirect impacts of pesticides through reduction of food supply may be important (Boatman et al., 2004; Goulson, 2014). The most definitive link was revealed after decades of research on grey partridges (*Perdix perdix*) in England, where declines were linked to a reduction in insect prey availability due to herbicide use (Southwood and Cross, 1969; Potts, 1986). Availability of insect prey can be altered directly through use of insecticides, or indirectly by herbicides or fertilizers affecting plant communities and reducing insect habitat. Sodium ammonium nitrate fertilizer sprayed in a pasture resulted in defoliated blackberry bushes and stunted bahia grass, decreasing prey availability, reproductive success, and survival for loggerhead shrikes (*Lanius ludovicianus*) (Yosef and Deyrup, 1998). A meta-analysis by Attwood et al. (2008) revealed that arthropod species richness was greater in areas of less intensive agriculture; maximum richness existed in native vegetation relative to agricultural land of any intensity and was lowest in fields with conventional cropping.

Agrochemical use has significant effects on arthropod biomass and abundance; experimental weedy plots contained 12 times more biomass of common insects eaten by nestling birds compared to monoculture plots prior to pesticide spraying, but following spraying with bromoxynil and imazamethabenz herbicides, weedy plots contained only 3 times more biomass than monoculture plots (Taylor et al., 2006). Organic farms, which do not use synthetic pesticides or fertilizers, harbour higher insect abundance compared to conventional farms (Hole et al., 2005). Furthermore, pesticides may alter the diet of insectivorous birds; using a 48-year dietary record in chimney swifts (*Chaetura pelagica*), the rise of DDT and its metabolites were correlated with a transition in diet from Coleoptera, which were greatly reduced due to DDT use, to Hemiptera (Nocera et al., 2012). Such a diet shift could result in

adverse nutritional consequences, as Coleoptera provide a greater caloric value (Kumar, 1996).

Agricultural practices can disrupt the natural emergence and availability of aerial insect prey throughout the breeding period. Paquette et al. (2013) reported that dipteran prey abundance, while similar in early spring, became progressively lower on agriculturally intensive sites during the breeding season. Reductions in prey availability can alter adult foraging behavior (Winkler et al., 2013; Schifferli et al., 2014; Stanton et al., 2016) with knock-on effects on reproduction and survival. Delayed effects of diminished food supply may be expected, with changes in bird populations being detected in the year following reduced prey abundances. Temporal delays would be expected if food availability impacted breeding success or post-fledging and overwinter survival (Benton et al., 2002).

5. Analysis of drivers of farmland bird declines

As reviewed above, multiple drivers and mechanisms can influence farmland bird population parameters such as reproduction and survival through both direct and indirect pathways (Fig. 1). Our systematic analysis of the frequency of farmland-associated bird declines from published studies confirmed the relative importance of two drivers: the single and combined effects of pesticides (often insecticides) and habitat loss. Of 122 unique studies investigating the effects of agriculture on farmland bird species (Appendix A), 51 (41.8%) reported negative effects from pesticides, 33 (27.1%) from habitat loss or fragmentation, 17 (13.9%) from mowing and harvesting operations, 11 (9.0%) from grazing disturbance, and 4 (3.3%) from reduced food availability (Fig. 4a; note the total percentage does not add to 100%, as some studies included data for multiple drivers). Within each species-specific habitat group (aerial, grassland, shrubland), aerial insectivores and shrubland species were most commonly affected by pesticides (63.6% of studies for aerial species, 69.8% for shrubland), while grassland species were equivalently influenced by habitat loss (37.0%) and pesticides (38.6%). By guild, physiology was the most commonly studied effect for aerial insectivores (45.5% of studies), abundance, density, or occurrence (42.1%) among grassland species, and survival (47.2%) among shrubland species.

Studies on the two most frequently studied drivers, pesticides and habitat loss, reported responses in 42 and 36 species, respectively, demonstrating that a wide variety of farmland bird species are affected (Fig. 4b). In contrast, 24 studies (19.7%) reporting no effect or a positive effect of agriculture encompassed 29 different species, most frequently clay-coloured ($n = 6$ studies) and savannah ($n = 7$) sparrows. These results were most commonly associated with grazing disturbance, where 42% ($n = 8$) of studies found positive or no effects. Of positive or no effect studies, 58% measured bird abundance, density, or occurrence (Fig. 4b). Within each driver category, reduced food availability was the most consistently negative, with 100% of those studies reporting negative effects, but the sample size was small ($n = 4$). Other driver categories including pesticides (93% negative) and mowing/harvesting (81% negative) also typically reported negative effects on farmland birds, including negative effects on survival (43% and 11%, respectively) and reproduction (26% and 37%), two factors strongly linked to population declines (Newton, 2004).

While total amount of applied insecticides (not all pesticides) and cases of acute lethal poisoning in birds have declined in recent decades (Köhler and Triebkorn, 2013; Osteen and Fernandez-Cornejo, 2013), increasing frequency and area of use, widespread seed-coating applications, and toxicity to birds remain problematic. This is coupled with the ongoing degradation of grassland and wetland habitat for agricultural use (Bartzen et al., 2010), which suggests losses of suitable farmland habitat, small scale habitat features and landscape complexity remain significant for many bird species in decline. A recent meta-analysis concluded that biodiversity losses were more strongly related to pesticide use than agricultural land area, although the authors

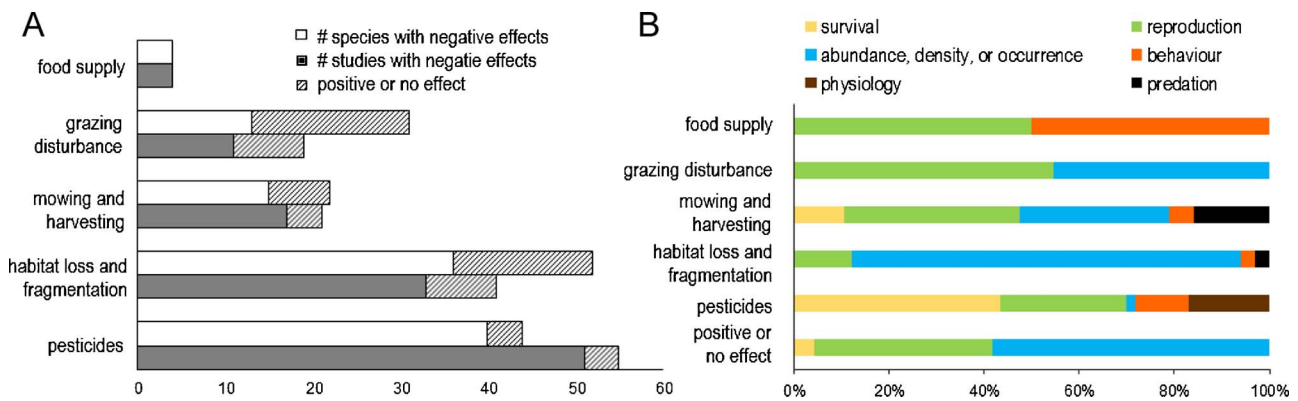


Fig. 4. Summary of the number of published studies ($n = 122$) and number of species ($n = 59$) for each primary agricultural driver that reported either a significant negative, positive or no effect ($p > 0.05$) (a). Percentage of studies investigating the different effect parameters (survival, reproduction, abundance, behaviour, physiology, and predation) for each of the 5 identified drivers or reporting a positive or no effect (b).

cautioned that pesticide use could be correlated with other causal factors that are a proxy for agricultural intensification (e.g. farm size, disturbance frequency or plant diversity) (Gibbs et al., 2009). Mineau and Whiteside (2013) found acute pesticide toxicity was the best predictor of grassland bird population declines. However, a re-analysis by Hill et al. (2014) determined that models for grassland bird declines that included habitat availability had 1.3–21.0 times more support than insecticide acute toxicity alone. Therefore, we conclude that both pesticide use and habitat loss appear to be major drivers of agriculturally-associated bird declines in North America, with their magnitude of influence differentially affecting species at multiple spatial scales.

6. Recommendations for farmland management

Given evidence reported here, management of intensive agricultural landscapes should focus on addressing the two most common drivers of farmland bird population declines – loss of habitat and pesticide use – while considering actions that are economically sustainable. Reversal of avian declines can occur through both restoration of native grassland as well as implementing beneficial farming practices (Askins et al., 2007). The US Conservation Reserve Program (CRP), which provides landowners with monetary incentives to convert highly erodible cropland into wildlife habitat, has positively influenced a variety of grassland birds across the continent (Reynolds, 2005; Blank, 2013; Haegen et al., 2015). Natural buffers in agricultural fields can also serve an important conservation role, with avian abundance, richness, and diversity all higher in fields with 10% of the area converted to strips of perennial native grass when compared to control fields (Schulte et al., 2016).

Therefore, we recommend re-establishment or preservation of field margins, grassland patches, and other uncultivated areas to develop a more complex and permanent vegetation structure. This would allow for higher plant and insect diversity, as well as more nesting and foraging habitat for farmland birds, and would likely have a minimal impact on farm revenue or operations. Well-maintained agricultural buffers and prairie strips can reduce runoff and increase abundance of pollinators and birds while enhancing soil and nutrient retention (Schulte et al., 2017); minimizing soil loss has been found to increase crop yields (Pimentel et al., 1995). Fields with edge habitat composed of diverse plants support higher diversity and abundance of bird species (Kross et al., 2016; Wilson et al., 2017); in turn, areas with higher avian abundance have reported lower densities of crop pests (e.g. weevils, a common crop pest on alfalfa) (Kross et al., 2016). In California, predation rates on insect pest species by farmland birds were highest in cropped areas near hedgerows and shrubby field margins (Garfinkel and Johnson, 2015). Planting sunflower (*Helianthus annuus*) intercrops resulted in greater abundance and foraging activity of insectivorous birds over the cropped area than in control plots (Jones and Sieving,

2006). Targeted management to increase use of field margins and habitat complexity on agricultural areas can benefit both farmers and biodiversity; despite these benefits, as of 2009 only 22.2% of farms in the US were enrolled in conservation programs supporting buffer use (Claassen et al., 2014). Obstacles and knowledge gaps inhibiting greater adoption of these programs need to be addressed including determining what the ideal spatial configuration is, what type of vegetation is most advantageous for various geographic regions and farmland birds and the associated costs.

We also recommend practices that support principles of integrated pest management (IPM) and, when feasible, organic or low-input systems, which can result in higher richness and abundance of a variety of organisms, including many bird species, compared to conventional, non-organic farming (Freemark and Kirk, 2001; Beecher et al., 2002; Hole et al., 2005; Kirk and Lindsay, 2017). It is thought that avian species dependent upon crops for foraging would particularly benefit from organic agriculture, as differences in nestling food biomass are more pronounced within crop fields compared to other agricultural areas (Girard et al., 2012). While this type of agriculture could be highly beneficial from a biodiversity viewpoint, a recent meta-analysis found that organic farming still results in 19.2% ($\pm 3.7\%$) lower yields than conventional farming (Ponisio et al., 2014), which could produce concerns for global food security with more widespread use. With the recent widespread application of highly toxic pesticides routinely used as seed dressings, such as with neonicotinoids and many fungicides, prophylactic use frequently leads to higher chemical inputs and farm costs than may be necessary. Fields using integrated pest management can lower costs to farmers and reduce pesticide risks as compared to prophylactic treatment, despite both methods producing an equivalent crop yield (Bueno et al., 2011). Concurrent monitoring of bird population responses and farm yields is a valid method to identify beneficial changes in farmland management that can be weighed against costs (e.g. Perlut et al., 2011; Evans et al., 2014).

7. Research needs

While farmland and grassland birds have been relatively understudied, the number of papers published on the topic more than doubled from 1985 to 2010 (Koper and Nudds, 2011). However, studies of the lethal and sublethal effects of pesticides in field settings are needed, as are those to determine whether agriculturally-mediated declines in food supply indirectly affect farmland birds. There are still few field and captive investigations into the ecotoxicity of newer and widely used insecticides such as neonicotinoids (Mineau and Palmer, 2013; Eng et al., 2017).

Studies have demonstrated negative relationships between agricultural intensification and insect availability, and reduced food availability and lower survival and reproduction in birds, but rarely

have these two topics been explicitly linked. Multi-trophic level linkages have been investigated for several species in Europe (see [Brickle et al., 2000](#); [Morris et al., 2005](#); [Hart et al., 2006](#)), but not in North America. There is also a need for research over larger spatial scales. Studies investigating the effectiveness of agri-environment schemes in promoting biodiversity in five European countries found mixed responses for different taxa and strongly recommend the incorporation of research over larger scales ([Kleijn et al., 2006](#)). Considerable variation in effects between regions exists, and so it is also imperative to determine the most effective conservation strategies that are locally adapted. Investigating the potential benefits of farmland improvement through adaptive management experiments would produce vital information, and help to determine biological and socioeconomic viability of different farm management schemes. Improved understanding of the mechanisms underpinning farmland bird declines and using this information to design multi-functional agricultural landscapes is urgently needed to stabilize or reverse the negative trends for many farmland-associated bird populations.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <https://doi.org/10.1016/j.agee.2017.11.028>.

A summary of species for which there have been reported effects of agriculture (Appendix A) is available online.

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