Western Specialty: Greater Sage-Grouse

The population of the Greater Sage-Grouse has declined dramatically because of multiple factors, of which one is large-scale extraction of fossil fuels from under its habitat. Besides the habitat loss, fragmentation, and disturbance resulting from this industry, the noise that drill rigs generate is also a factor. In this issue of *Western Birds*, Skip Ambrose, Christine Florian, Justin Olnes, John MacDonald, and Therese Hartman evaluate the best methods for quantifying noise pollution in the sagebrush steppe of Wyoming and evaluate its effects on the Greater Sage-Grouse through 20 years of monitoring at leks. They confirmed that noise has an adverse effect on the grouse and identified the threshold above which the effect is significant.
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The 45th Annual Report of the California Bird Records Committee: 2019
Records  Thomas A. Benson, Deborah J. House, Guy McCaskie, Alex M. Rinkert, Adam J. Searcy, and Ryan S. Terrill .............................................. 2

Sagebrush Soundscapes and the Effects of Gas-Field Sounds on Greater Sage-Grouse  Skip Ambrose, Christine Florian, Justin Olnes, John MacDonald, and Therese Hartman ..................................................... 23

The Status of Loons in Idaho  Daniel M. Taylor ............................................ 47

Monitoring through Community Science: Anna’s Hummingbird Winter Range Expansion into Idaho  Jessica J. Pollock, Heidi Ware Carlisle, Heather M. Hayes, and Bryce W. Robinson .................. 58

Evidence of Brood Parasitism and Quantification of Rangewide Overlap between the Olive Warbler and Brown-headed Cowbird Jessie L. Williamson and Matthew J. Baumann ........................................... 68

NOTES

First Documentation of Successful Breeding for the Anna’s Hummingbird in Idaho Bryce W. Robinson, Jessica J. Pollock, Heidi Ware Carlisle, Heather M. Hayes, and Janice Engle ............................................................. 76

Apparent Commensalism of a Red-tailed Hawk and Badger James W. Cornett ............................................................................................ 80

Book Review: Breeding Bird Atlas of Nevada County, California W. David Shuford ........................................................................................... 82

Featured Photo: Do Nutcrackers Use Automobiles as Nutcrackers? P. Ross Gorman ......................................................................................... 87

Thank You to Our Supporters ........................................................................... 90

Front cover photo by © Larry Sansone of Los Angeles, California: Spotted Redshank, near Ormond Beach, Oxnard, Ventura Co., California, 15 September 2019.

Back cover “Featured Photo” by © P. Ross Gorman of Cody, Wyoming: Clark’s Nutcracker placing a salamander on its back in the middle of an active roadway, 22 April 2019, Yellowstone National Park, Wyoming. Note on both the salamander and the nutcracker the whitish droplets of toxin excreted by the salamander, suggesting that the salamander’s skin had already been pinched or pierced. But the salamander is otherwise intact and actively defending itself.

Western Birds solicits papers that are both useful to and understandable by amateur field ornithologists and also contribute significantly to scientific literature. Particularly desired are reports of studies done in or bearing on North America west of the 100th meridian, including Alaska and Hawaii, northwestern Mexico, and the northeastern Pacific Ocean.

Send manuscripts to Daniel D. Gibson, P. O. Box 155, Ester, AK 99725; avesalaska@gmail.com. For matters of style consult the Suggestions to Contributors to Western Birds (at https://westernfieldornithologists.org/publications/journal).
THE 45TH ANNUAL REPORT OF THE CALIFORNIA BIRD RECORDS COMMITTEE: 2019 RECORDS

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ABSTRACT: From its last report through 2019, the California Bird Records Committee reached decisions on 204 records involving 225 individuals of 62 species and one species group, endorsing 176 records of 197 individuals. These include the first accepted state record of the Yellow-browed Warbler (Phylloscopus inornatus), bringing California’s total list of accepted species to 676, of which 13 represent established introductions. Other especially notable records detailed in this report include the state’s third Common Eider (Somateria mollissima), third Common Ringed Plover (Charadrius hiaticula), fourth and fifth Red-flanked Bluetail (Tarsiger cyanurus), a significant incursion of Cassin’s Sparrows (Pooeaeae cassinii) into the eastern Mojave Desert, and the first Spotted Redshank (Tringa erythropus) in 30 years.

This 45th report of the California Bird Records Committee (CBRC), a committee of Western Field Ornithologists, summarizes evaluations of 204 records involving 225 individuals of 62 species and one species group. The committee accepted 176 of the 204 records, involving 197 individuals of 62 species and one species group, for an acceptance rate of 86.3%. A record is considered accepted if it receives no more than one “not accept” vote from the nine voting members on the grounds of questionable identification, or no more than two “not accept” votes on the grounds of questionable natural occurrence. We consider 24 records of 16 individuals to represent returning or continuing birds that were accepted previously. Twenty-eight records, involving 28 individuals of 22 species, were not accepted because the identification was not considered to be substantiated. For review, reports of
multiple individuals together are given the same record number; we report the total number of accepted individuals, which may be different from the number of accepted records. Most of the records in this report are of birds first documented in 2019, although a few are earlier.

Following the recent split of the Mexican Duck (Anas diazi) from the Mallard (A. platyrhynchos) by the American Ornithological Society (Chesser et al. 2020), the committee will be evaluating the addition of the Mexican Duck to the California list on the basis of several reports still pending review and will publish the results of its analysis in a future report.

Species-account headings are organized with English and scientific names first, followed in parentheses by the total number of individuals accepted for California (this report included) and the number of new individuals accepted in this report. Accounts summarize records accepted (as applicable), followed by records not accepted because the identification was not established, the date or location was uncertain, or the natural occurrence was questionable (as applicable). A double asterisk (**) following the number of accepted individuals indicates that the species has been reviewed for only a restricted period, so the number of accepted individuals does not represent the total number for the state. When the observer(s) who originally discovered the bird provided documentation, their initials are listed first in italics, followed by the initials of subsequent observers supplying documentation. A symbol following an observer’s initials indicates she or he submitted a photograph (†), sketch ($), audio recording (§), and/or video (‡). The absence of a symbol following the observer’s initials indicates the submission of written documentation only. After the observers’ initials is the record’s identifying number assigned by the CBRC’s secretary. A (#) precedes a specimen number; we cite the collections of the Natural History Museum of Los Angeles County (LACM) and San Diego Natural History Museum (SDNHM) in this report.

As of the CBRC’s 43rd report (Singer et al. 2020), age terminology follows that of Humphrey and Parkes (1959) as modified by Howell et al. (2003) and Howell and Pyle (2015). Definitions of abbreviations and additional details regarding minutiae of formatting may be found in the CBRC’s previous annual reports and in CBRC (2007), both available via the CBRC’s website: www.californiabirds.org. Also available through this website are the California bird list, the review list, an online form for submitting documentation of review species, committee news, recent photos of rare birds, the CBRC’s bylaws, and a form for querying the CBRC’s database. Observers are encouraged to submit documentation for all species on the CBRC’s review list to the secretary via e-mail (secretary@californiabirds.org) or the website. Documentation of all records is archived at the Western Foundation of Vertebrate Zoology (www.wfvz.org) and is available for public review by appointment or by contacting the CBRC’s secretary.

**GARGANEY** Spatula querquedula (28, 4). A male in West Sacramento, YOL, 1–17 Mar 2019 (GE†, LC†, JAH†, NJO†, LPi†, MJR†, CNR, JCS†, RSt†, LST†, SBT†; 2019-021) may have wintered locally, and a female at New Chicago Marsh at the south end of San Francisco Bay, SCL, 7 Jun 2019 (AH†; 2019-051) was evidently a spring migrant. A fall migrant undergoing preformative molt was with other teal in Baker, SBE, 20 Sep 2019 (JEP†; 2019-124) and represented a first record for San Bernardino County. A first-year female associating with Blue-winged Teal (S. dis-
spent the winter at the Salton Sea State Recreation Area, RIV, 10 Nov 2019–30 Mar 2020 (TAbet†, CAD†, MAG†, AHit†, DJH†, PAI†, GM, RLM†, SJM†, ES, AJSt†, DSt†, JTS†; 2019-174; Figure 1).

After California’s first Garganey in Long Beach, LA, on 19 Mar 1975 (Luther et al. 1979), 22 more were recorded in the following 27 years, followed by four years with none. This pattern supports comments by Howell et al. (2014) that records in western North America have declined markedly since the late 1990s. However, an additional four have been recorded since one spent the winter of 2017–2018 in Santa Maria, SBA (Singer et al. 2020), and two more individuals were reported in 2020 (CBRC 2020).

**COMMON POCHARD* Aythya ferina** (5, 1). An adult male on San Pablo Bay, SOL, was shot by an anonymous hunter on 9 Jan 2019 (†, *fide* DSS; 2019-003); the specimen was not preserved. Although five Common Pochards have now been found in California, there is only one other report of this Eurasian duck in North America south of Alaska, from Victoria, British Columbia, in November 2020 (www.eBird.org).

**KING EIDER* Somateria spectabilis** (46, 1). A female in formative plumage was at the Smith River mouth, DN, 27 Nov–16 Dec 2019 (BT†, DK-B†; 2019-190).

**COMMON EIDER* Somateria mollissima** (3, 1). An adult female at Pebble Beach, MTY, 12–28 Feb 2019 (TAbet†, PEL, GM, OTM†, NJO†, DRob†, CNR, NKSt†, LST†, SBT†; 2019-016; Figure 2) represents the southernmost record for the Pacific coast. This individual showed the narrow, pointed frontal lobes of the bill that are characteristic of the subspecies *v-nigrum* (see Goudie et al. 2000). The previous two Common Eiders in California were both in Del Norte County, the first in 2004 (*S. m. v-nigrum*; Cole et al. 2006), the second in 2011 (*S. m. dresseri*; Nelson et al. 2013, Able et al. 2014). Far fewer Common than King Eiders have been reported along the west coast from southern British Columbia south: fewer than a dozen of the former, while the latter has occurred annually.

**RUDDY GROUND DOVE* Columbina talpacoti** (118**, 1). A male was with Inca Doves (*C. inca*) at Oasis, RIV, 2 Oct 2019 (RLMS; 2019-132).

**RUBY-THROATED HUMMINGBIRD* Archilochus colubris** (21, 2). Females in juvenile or formative plumage on San Clemente Island, LA, 5 Sep 2014 (JTS†; 2014-186), and at the El Carmelo Cemetery in Pacific Grove, MTY, 9–12 Sep 2019 (PBA†, CR†, JeB†, BH†; 2019-204) provided the first records for Los Angeles County and Monterey County, respectively.

**BROAD-BILLED HUMMINGBIRD* Cynanthus latirostris** (112, 3). A female in formative plumage frequented a feeder in Lone Pine, INY, 15–17 Apr 2019 (RK, MP†; 2019-049); only 10 of California’s 112 Broad-billed Hummingbirds have occurred between April and July. An adult male remained near Monastery Beach, Carmel, MTY, 24 Oct–18 Nov 2019 (RC†, TD†, BH†, DRob†; 2019-165). The committee also endorsed a belated report of an adult male photographed at the Living Desert Zoo and Gardens, Palm Desert, RIV, 27 Nov 2010 (AG†; 2010-197).

**COMMON RINGED PLOver* Charadrius hiaticula** (3, 1). An adult male at the San Joaquin Wildlife Sanctuary, Irvine, ORA, 28 Aug–3 Sep 2019 (BAr†, JMB†, CHar†, TABet†, SC†, GM, MMet†, NJO†, ES†, SBT†, CY†; 2019-099) provided the third record for California. Elsewhere on the Pacific coast south of Alaska, the only records are of single fall migrants in British Columbia at Oyster Bay in 2018 (https://bcfo.ca/bc-bird-records-committee-sightings-database/) and Boundary Bay in 2019 (www.eBird.org), and in Washington at Port Susan Bay in 2006 (Aanerud 2011).
UPLAND SANDPIPER *Bartramia longicauda* (36, 2). A juvenile was at the Silver Saddle Resort, Galileo Hill, KER, 6–7 Sep 2019 (KO†, TABe†, CHO, RH; 2019-103), and one of unknown age was at Elings Park, Santa Barbara, SBA, 8 Sep 2019 (PAG; 2019-104). In California, fall migrants have been recorded between 8 August and 28 October, with 75% of these between 17 August and 23 September.

BAR-TAILED GODWIT *Limosa lapponica* (54, 3). One in formative plumage was at Oro Loma Marsh along the Hayward Regional Shoreline, ALA, 21–25 Jul 2019 (MJ†, DRot†, JT†, BT†, MPe†; 2019-066). Juveniles were at Tunitas Creek Beach, SM, 16–24 Oct 2019 (DA†, WGB†, MDPe†, CHay†, RH†, JMo†, NJO†, LPi†; 2019-146) and Emeryville Marina, ALA, 31 Oct–6 Dec 2019 (SJ†, LB†, JMo†, MJR†; 2019-159).

HUDSONIAN GODWIT *Limosa haemastica* (59, 4). An adult male in alternate plumage at the San Diego River mouth and adjacent Mission Beach, SD, 27 May–2 Jun 2019 (DWA†, DBat†, TABe†, NJD†, TGu†, GM, JRM†, NJO†, GLR, JTS†; 2019-038) was the first Hudsonian Godwit recorded in San Diego County. Fall migrants in juvenile plumage included one at the Salinas River National Wildlife Refuge (NWR), MTY, 1–7 Aug 2019 (RC†, DRob†, CR†, RAR†, SBt†; 2019-077); another at the Aracata Marsh, HUM, 24 Aug 2019 (TK†, MVB†; 2019-087); and a third at Tunitas Creek Beach, SM, 14 Oct–16 Nov 2019 (DSid†, DA†, WGB†, MDPe†, JG†, CHay†, RH†, JMo†, NJO†, AP†, LPi†; 2019-141). IDENTIFICATION NOT ESTABLISHED: The report of one at El Segundo Beach, LA, 2 Apr 2019 (2019-055) received no support.

RED-NECKED STINT *Calidris ruficollis* (26, 1). One in alternate plumage was at the San Pablo Bay NWR, SOL, 7 Aug 2019 (RM†; 2019-081).

LITTLE STINT *Calidris minuta* (34, 0). The committee infers that one at the south end of San Diego Bay, SD, 2 Aug 2019–3 Apr 2020 (PEL, CA†, NC†, NF†, MAG†, DJ, GM, AN†; 2019-080) was the same bird as at that site 20 Oct 2018–24 Apr 2019 (2018-219; Benson et al. 2020).

SPOTTED REDSHANK *Tringa erythropus* (6, 1). A juvenile remained near Oxnard, VEN, 15–25 Sep 2019 (LSa†, TABe†, GM, DPe†, AJS†, RST†, SBT†; 2019-119; see this issue's front cover). It had begun the preformative molt, having replaced a few of the juvenile scapulars with unmarked pale gray feathers. California's five previous records fall between 1983 and 1989 (CBRC 2007), making this the state's first Spotted Redshank in 30 years.

LONG-BILLED MURRELET *Brachyramphus perdix* (31, 0). IDENTIFICATION NOT ESTABLISHED: Photos of a black-and-white murrelet off Centerville Beach, HUM, 15 Jul 2004 (2004-224) did not show the expected white scapular stripes and failed to convince five committee members that it was this species. There have been no records of the Long-billed Murrelet in California since 2015.

BLACK-HEADED GULL *Chroicocephalus ridibundus* (31, 1). The committee considers an adult in basic plumage at the North Shore Marina, Salton Sea, RIV, 3 Jan 2019 (DTR†; 2019-002) and 13 km to the west at Oasis, RIV, 13 Feb–16 Mar 2019 (RLM†, MAG†, CMcG†; 2019-017) to be the same bird first reported frequenting the same region during the winter of 2013–2014 (Singer et al. 2016), then every winter since the winter of 2016–2017 (Singer et al. 2020). An adult in alternate plumage was at Bouldin Island, SJ, 22 Feb 2019 (DFD; 2019-020).

BLACK-TAILED GULL *Larus crassirostris* (8, 2). The committee considered an adult in alternate plumage at the mouth of Gazos Creek, SM, 30 May 2019 (MDFe†, CHay†; 2019-041) to be the same as one on Southeast Farallon Island, SF, 23 Oct–1 Nov 2019 (JRT†; 2019-150).

SLATY-BACKED GULL *Larus schistisagus* (72, 4). One in its second cycle was at Point Pinos, MTY, 1 Jan 2019 (BTM†; 2019-001). Single adults were on the coast at
Gazos Creek Beach, SM, 21 Jan 2019 (AJ†; 2019-005); at Marina State Beach, MTY, 29 Jan–15 Mar 2019 (BTM†; 2019-009); and well inland at the Koster Road gravel pit near Tracy, SJ, and the nearby Delta–Mendota Canal near Vernalis, STA, 17–23 Feb 2019 (JG†, JHH†, JCS; 2019-019). This Asiatic gull is now recorded in all but four of the coastal counties from Del Norte to Los Angeles, around San Francisco Bay, and inland around the Sacramento/San Joaquin River delta (CBRC 2007, 2020). IDENTIFICATION NOT ESTABLISHED: Reports of first-cycle Slaty-backed Gulls at the Koster Road gravel pit near Tracy, SJ, 24 Feb 2019 (2019-023) and at Point Pinos, MTY, 28 Dec 2019 (2019-209) received little support from committee members.


WEDGE-TAILED SHEARWATER Ardenna pacifica (11, 0). IDENTIFICATION NOT ESTABLISHED: The report of one seen briefly during a pelagic birding trip ~35 km south-southeast of Southeast Farallon Island, SF, 29 Sep 2019 (2019-130) lacked sufficient documentation.

MASKED BOOBY Sula dactylatra (46, 6). Masked Boobies continued to occur in above-average numbers for the fifth consecutive year, though the six in 2019 were fewer than the 2018 peak of 14 (Benson et al. 2020). All records in 2019 were from southern California, where 80% of the state’s Masked Boobies have occurred. Single adults flew past Point La Jolla, SD, 3 Feb 2019 (JD; 2019-012) and Cabrillo National Monument, SD, 31 Aug 2019 (GN†; 2019-098). A long-staying adult, first seen ~18 km southeast of Point Fermin, LA, 19 Jul 2019 (GG†; 2019-065), was subsequently encountered ~3 km outside Los Angeles harbor, 25 Jul 2019 (MW†; 2019-072), at Dockweiler State Beach, LA, 9 Aug 2019 (DSc†; 2019-082), at Hollywood Beach in Oxnard, VEN, 12 Aug 2019 (JMa†; 2019-086), and offshore in the San Pedro Channel, LA, 25 Aug 2019 (GG†, BAl; 2019-093). Three individuals in their second cycle occurred in November and December 2019. One was discovered on the Oceanside pier, SD, 21 Nov 2019 (MA†, EB†; 2019-198), when it was captured for rehabilitation at Sea World. It was released off La Jolla 10 Dec 2019 and subsequently seen nearby at Point Loma on 12 December and on the Zuñiga Jetty at the mouth of San Diego Bay on 14 December. The second was found on Dog Beach in Huntington Beach, ORA, 18 Dec 2019 (LPe†; 2019-202) and taken in for rehabilitation at the Wetlands and Wildlife Care Center. The third was seen on Moonlight State Beach, Encinitas, SD, 30–31 Dec 2019 (JMM†, MBi†; 2019-203). The committee also endorsed the report of a second-cycle Masked Booby ~17 km north-northwest of Santa Catalina Island, LA, 16 Sep 2018 (MiF†; 2018-246). IDENTIFICATION NOT ESTABLISHED: Documentation for two reports, 2019-096, and 2019-100, did not eliminate the Nazca Booby; see records 2019-096A and 2019-100A, respectively, in the Masked/Nazca Booby account for more details.

NAZCA BOOBY Sula granti (62, 21). In 2019, Nazca Boobies were recorded in double-digit numbers for the second year in a row, with most occurring off the southern California coast. Single adults were seen ~9.4 km south-southwest of Dana Point, ORA, 11 Jun 2019 (KMC†; 2019-053); ~177 km west-southwest of San Nicolas Island, VEN, 19 Jul 2019 (CW†; 2019-091); ~2.7 km south-southwest of the Newport Beach pier, ORA, 24 Jul 2019 (KMC†; 2019-070); ~7 km west of Point Conception, SBA, 25 Jul 2019 (CW†; 2019-075); at Point La Jolla, SD, 28 Jul–1 Aug 2019 (PEL, BLC; 2019-074); ~48 km west of Point Loma, SD, 18 Aug 2019 (GM, BRId†, JTS†; 2019-084); over the Thirty Mile Bank, SD, 2 Sep 2019 (AJS†; 2019-105); ~4 km west-northwest of Santa Barbara Island, SBA, 3 Sep 2019 (AJS†; 2019-106); ~109 km west-southwest of San Nicolas Island, VEN, 4 Sep 2019 (AJS†; 2019-107); and
Figure 1. This female Garganey in formative plumage, photographed 30 Nov 2019 at the Salton Sea State Recreation Area, Riverside County (2019-174), nicely shows the striped head and slightly smaller size that differentiate it from the similar female Blue-winged Teal behind it.

Photo by Thomas A. Benson

Figure 2. This Common Eider, an adult female photographed 17 Feb 2019 at Pebble Beach, Monterey County (2019-016), is only the third known to have reached California. It shows the sloping forehead and barred sides that differentiate it from a similar-looking female King Eider.

Photo by Brad K. Schram
over the Lasuen Seamount, ~23 km south-southwest of Newport Beach, ORA, 4 Oct 2019 (RCL†; 2019-134). A group of three adults flew out of San Diego Bay, SD, 6 Sep 2019 (AJS, JWe†; 2019-108). Two adults found on Santa Barbara Island, SBA, 12 Oct 2019 (NL, CSt†; 2019-145) were still there and observed allopreening 14 Nov 2019 (JHo†; 2019-179). One undergoing its second prebasic molt, first photographed at Camp Pendleton, SD, 23 Jul 2019 (DBi†; 2019-069), was subsequently seen offshore from San Clemente and at Dana Point Harbor, ORA, 25 Jul–15 Aug 2019 (FB†, JLD, BEE†, TGu†, DK-B†, ES†; 2019-071). Two additional birds undergoing their second prebasic molt were ~7 km southeast of Ventura Harbor, VEN, 3 Aug 2019 (JoB†; 2019-079) and in Los Angeles Harbor, LA, 17–19 Nov 2019 (KM†, CLY†; 2019-178). Individuals undergoing their third prebasic molt were over the Nine Mile Bank, SD, 18 Aug 2019 (MV†; 2019-089) and 13 Oct 2019 (NC†, GM; 2019-139), while a third found at Naval Air Station North Island, SD, 26 Aug 2019 (JWh†, LDS†, PU; 2019-097) was taken into rehabilitation at Sea World but later perished (SDNHM #56365). There were only two records from northern California in 2019: an adult at Southeast Farallon Island, SF, 2 Feb 2019 (JS†, PW†; 2019-011) and one undergoing its second prebasic molt over Cabrillo Canyon, SCZ, 5 Oct 2019 (AMR†; 2019-136).

IDENTIFICATION NOT ESTABLISHED: Documentation submitted for one report, 2019-095, did not eliminate the Masked Booby; see record 2019-095A in the Masked/Nazca Booby account for more details. Since accepting the first California record of the Nazca Booby, of one found dead at McGrath State Beach, VEN, 23 Jul 2013 (Rottenborn et al. 2016), the committee has now accepted 62 records for an average of 8.9 per year during the 7-year period from 2013 to 2019. As a result, at its annual meeting in 2020 the committee voted to remove this species from the review list; we review records of the Nazca Booby through 2019 only.

MASKED/NAZCA BOOBY Sula dactylatra/granti (48, 7). The following reports of black-and-white boobies lacked documentation sufficient for identification to species, but the committee endorsed them as Masked/Nazca Boobies: an adult or near-adult at South Cardiff State Beach, SD, 5 Feb 2019 (RTP†; 2019-013); an adult ~2.7 km west of the Las Pulgas Creek mouth, Camp Pendleton, SD, 28 Jun 2019 (SM†; 2019-059); individuals undergoing their second or third prebasic molt flying past Imperial Beach, SD, 29 Jun 2019 (MSad†, JTS; 2019-063) and past Point La Jolla, SD, 24 Aug 2019 (DJ, SW; 2019-095A); one undergoing its second prebasic molt ~5 km offshore from Dana Point, ORA, 23 Aug 2019 (RCL†; 2019-094); and an adult (BAL, MES†; 2019-096A) and one in second prebasic molt (BAL†, MES†; 2019-100A) together on Santa Barbara Island, SBA, 24 Aug 2019. The committee reviews records of the Masked/Nazca Booby through 2019.

RED-FOOTED BOOBY Sula sula (85, 18). For the second consecutive year, Red-footed Boobies staged a significant incursion into the waters off southern California, though the total number of individuals was only half of that in 2018. All records below are of brown-morph birds undergoing their second prebasic molt unless noted otherwise. Two individuals in their first or second plumage cycle flew past the Huntington Beach pier, ORA, 1 Jan 2019 (BED; 2019-022). One on Platform Edith, 14 km off Huntington Beach, ORA, 9 Feb 2019 (TAbet; 2019-014) was still there 4–11 May 2019 (TAbet†, TGu†, DH†, ES†; 2019-033), with two more 2 km southeast at Platform Eureka, ORA, 4 May 2019 (TAbet†, TGu†, DH†, DET†, ES†; 2019-034A and 2019-034B). Another was ~2.3 km west-southwest of the Newport Beach pier, ORA, 29 May 2019 (KMC†; 2019-040). Following this cluster of sightings in May, later in 2019 most of the Red-footed Boobies occurred between July and October: in Avalon Harbor, Santa Catalina Island, LA, 14–15 Jul 2019 (CB†; 2019-064); near Santa Catalina Island, LA, 21–23 Jul 2019 (CSel†; 2019-068); ~4 km southwest of Crescent Bay Point Park, Laguna Beach, ORA, 30 Jul 2019 (KMC†; 2019-076); ~9 km southwest of the Huntington Beach pier, ORA, 13 Aug 2019 (DC†; 2019-109);
over the Thirty Mile Bank, SD, 18 Aug 2019 (adult; GM, JTS†; 2019-085); ~8 km south-southwest of Dana Point, ORA, 23 Sep 2019 (undergoing third prebasic molt; CMo†; 2019-121); and ~17 km south of Goleta Point, SBA, 26 Oct 2019 (RPe†; 2019-164). One found on the beach at Coronado, SD, 14 Sep 2019 (AME†, PU†; 2019-133) was captured for rehabilitation at Sea World but did not survive (SDNHM #56382). The final two birds recorded in southern California in 2019 were single juveniles ~16 km west-southwest of La Jolla, SD, 20 Oct 2019 (DF†; 2019-188) and on the jetty at the entrance to Newport Harbor, ORA, 16 Nov–17 Dec 2019 (TABe†, PAI†, DSie†; 2019-177). In northern California, one was on Southeast Farallon Island, SF, 23 Oct–27 Nov 2019 (KO†, MSi†, JRT†; 2019-151) and another (undergoing its third prebasic molt) was at Crissy Field, SF, 8 Nov–25 Dec 2019 (DK-B†, JMOf, LPt†, PP†, MJR†, JWat†, ANW†; 2019-169). In 2020 the committee voted to remove the Red-footed Booby from the review list; we review records through 2019 only.

TRICOLORED HERON Egretta tricolor (81**, 6). One in formative plumage was at Colorado Lagoon, Long Beach, LA, 6 Oct 2019 (RAH†; 2019-135). An adult at Rodeo Lagoon, MRN, 10–23 Nov 2019 was also observed flying past Crissy Field, SF, 11 Nov 2019 (WL, JMOf†, AP†, LPt†, MJR†, DSS†, LS†, JWa†, ANW†; 2019-170), representing the first and second records for San Francisco County and Marin County, respectively. Orange County hosted three birds, including two undergoing preformative molt or in formative plumage at San Joaquin Wildlife Sanctuary, ORA, 15 Oct 2019 (JAK†; 2019-140) and at Upper Newport Bay, ORA, 10 Nov 2019–18 Apr 2020 (TABe†, AB†, AC†, TGut†, MK†, B&KK†, RPh††, ES†, L-SV†; 2019-175), and an adult at Bolsa Chica Ecological Reserve, ORA, 26–28 Oct 2019 (CHA†, ES†, LW†; 2019-154). One undergoing its preformative molt was on San Diego Bay at Bayside Park and Sweetwater Marsh, Chula Vista, SD, 30 Oct–21 Dec 2019 (KR†, AN; 2019-158; Figure 3).

GLOSSY IBIS Plegadis falcinellus (40, 0). IDENTIFICATION NOT ESTABLISHED: Documentation for ibises reported as the Glossy at the Woodland–Davis Clean Water Agency, Woodland, YOL, 19–22 May 2019 (2019-036) and in the Prado Basin, RIV, 1 Apr–21 May 2019 (2019-078) did not eliminate the possibility of hybrids.

ROSEATE SPOONBILL Platalea ajaja (147, 1). One in its first cycle flew over Unit 1 of the Salton Sea NWR, IMP, 11 Oct 2019 (GM; 2019-138).

BLACK VULTURE Coragyps atratus (11, 0). The committee concluded that all records of the Black Vulture accepted in 2019 represented continuing or returning birds. Those pertaining to the one in the San Francisco Bay area since March 2014 (2014-027 et seq.; Singer et al. 2016, CBRC 2020) included reports from Point Reyes Station, MRN, 2 Apr 2019 (RSC; 2019-024); Inverness, Olema, and Nicks Cove, MRN, 8 Jun–21 Jul 2019 (CC†, KF†; 2019-050); Bodega Bay, SON, 24 Sep–9 Oct 2019 (EH†, JV†; 2019-122); and Bolinas Lagoon, MRN, 3–17 Nov 2019 (AHi†; 2019-171). An adult with an identifiable injury to its left eye near Bishop, INY, 12 Dec 2019–21 Feb 2020 (ND†, DJH†, RH†; 2019-197) is considered the same as the one in that area the previous winter (2018-225; Benson et al. 2020). In both winters it was molting its remiges. Like the Turkey Vulture (Chandler et al. 2010), the Black Vulture normally molts its remiges annually in May and June (P. Pyle pers. comm.). This anomalous molt timing might suggest prior captivity, leading one committee member to question whether the bird's occurrences were natural. IDENTIFICATION NOT ESTABLISHED: A report from Eureka, HUM, 3 May 2019 (2019-044) lacked sufficient documentation.

MISSISSIPPI KITE Ictinia mississippiensis (53, 0). IDENTIFICATION NOT ESTABLISHED: A report of one near University City, San Diego, SD, 14 Sep 2019 (2019-120) provided too few details to rule out other species.
Figure 3. This first-cycle Tricolored Heron was photographed 3 Nov 2019 at Sweetwater Marsh, San Diego Bay, San Diego County (2019-158).

*Photo by Andrew Newmark*

Figure 4. This dark juvenile Gyrfalcon near McArthur on 6 Dec 2019 (2019-205) represented a second Shasta County record.

*Photo by Josh Able*
Figure 5. This Sulphur-bellied Flycatcher, California’s 21st, was photographed 16 Sep 2019 at Point Lobos State Reserve, Monterey County (2019-117). The smaller and mostly dark bill, whitish eyebrow and mustache, and bold black malar differentiate it from the similar Streaked Flycatcher (*Myiodynastes maculatus*), a potential vagrant to California. The narrow shaft streak to the outer rectrix indicates a juvenile feather.

*Photo by Judith Ellyson*

Figure 6. California’s 19th Blue Jay, photographed 14 Nov 2019 at Fortuna, Humboldt County (2019-172).

*Photo by Rob Fowler*
COMMON BLACK HAWK *Buteogallus anthracinus* (12, 1). An adult was photographed at Dos Palmas Preserve, RIV, 11 Sep 2019 (SSt†; 2019-113). Most of California’s Common Black Hawks have occurred in spring; this record is only the third of a fall migrant.

BOREAL OWL *Aegolius funereus* (0, 0). IDENTIFICATION NOT ESTABLISHED: The report of one calling in Citrus Heights near Sacramento, SAC, 1 Jul 2019 (2019-060) did not receive any support from committee members. There are no accepted records of this species in California.

GYRFALCON *Falco rusticolus* (17, 1). A juvenile near McArthur, SHA, 6 Dec 2019 (JAb†, KPA; 2019-205; Figure 4) was only the second recorded for Shasta County, the previous Gyr falcon also occurring near McArthur almost 32 years earlier, on 26 Dec 1987 (1988-296; Pyle and McCaskie 1992).

GREAT CRESTED FLYCATCHER *Myiarchus crinitus* (64, 2). Individuals having suspended the preformative molt were at Laguna Grande Park, MTY, 28 Sep–1 Oct 2019 (JeB†, RC†, DRob†, CR†, BLS†; 2019-128) and in Birchim Canyon, INY, 21 Oct 2019 (NJO†, RO†, DPa†, JPa†, SBr†, JHei, TH, CHo†, RH†; 2019-147).

SULPHUR-BELLIED FLYCATCHER *Myiodynastes luteiventris* (21, 1). One in formative plumage was found at Point Lobos State Reserve, MTY, 16 Sep 2019 (RiF, JE†, PWF†; 2019-117; Figure 5).


YELLOW-BELLIED FLYCATCHER *Empidonax flaviventris* (30, 1). One having suspended the preformative molt was captured and banded on Southeast Farallon Island, SF, 13 Sep 2019 (JFGa†, JRT: 2019-118).

WHITE-EYED VIREO *Vireo griseus* (88, 2). A singing male was near Inverness, MRN, 31 May 2019 (DSS§; 2019-042), and a singing male in formative plumage was at Arcata Marsh, HUM, 21–23 Jun 2019 (RoF§, DK-B§, TK§; 2019-057).

BLUE-HEADED VIREO *Vireo solitarius* (89, 1). A male in formative plumage was at Fort Rosecrans National Cemetery, Point Loma, SD, 28–29 Sep 2019 (NC†, SD†, NF†, LWi†; 2019-129). IDENTIFICATION NOT ESTABLISHED: The documentation for an adult vireo reported as the Blue-headed at Huntington Central Park, Huntington Beach, ORA, 21 Sep 2019 (2019-123) did not rule out a bright Cassin’s Vireo (*V. cassinii*).

BLUE JAY *Cyanocitta cristata* (19, 1). One in formative plumage spent the winter in Fortuna, HUM, 8 Nov 2019–29 Feb 2020 (RoF†, DLe§, DK-B; 2019-172; Figure 6).

WINTER WREN *Troglodytes hiemalis* (27, 1). One at Putah Creek, SOL, 24–28 Oct 2019 (RM§, MarS†; 2019-156) was the sixth recorded in the Central Valley and the first in Solano County. IDENTIFICATION NOT ESTABLISHED: The spectrogram of the call of a supposed Winter Wren recorded at Clear Lake State Park, LAK, 29 Nov–30 Dec 2019 (2019-207) did not show the harmonic bands characteristic of this species (see Hejl et al. 2002).
YELLOW-BROWED WARBLER *Phylloscopus inornatus* (1, 1). One in formative plumage photographed at Markleeville, ALP, 24–25 Oct 2019 (TE†, TABe†, CHo†, RH†, GM, MMc†, AP†, DRob†; 2019-152; Figure 7) represented a first record for California. This individual showed the indistinct pale median crown stripe, relatively dull lores, distinct upper wing bar, and overall bright, contrasting plumage that help distinguish the Yellow-browed from the most similar *Phylloscopus* warblers (Alström and Olsson 1988, Lehman 2000, Howell et al. 2014), of which only Pallas’s Leaf Warbler (*P. proregulus*) has been recorded in northeast Asia (Brazil 2009) or North America (Lehman and Rosenberg 2007). The Yellow-browed Warbler breeds in the eastern Palearctic Region and winters primarily in southeast Asia (Clement 2020). Although detected increasingly as a fall vagrant in western Europe, this species rarely reaches North America (Howell et al. 2014). Previous records south of Alaska are of one in late fall in British Columbia (also in October 2019; https://bcfo.ca/brc-round 27-jan-feb-2020-accepted-records/) and one in early spring that likely wintered in Baja California Sur (Mlodinow and Radamaker 2007). Additionally, a single-observer sight record from Wisconsin (Frank 2007) has been placed on the Wisconsin Society for Ornithology’s checklist as a hypothetical species (https://wsobirds.org/images/pdfs/WI_Taxonomic_Order_2019.pdf).

DUSKY WARBLER *Phylloscopus fuscatus* (21, 1). One was at the mouth of Mal Paso Creek, MTY, 27–28 Oct 2019 (DRob†; 2019-180). While 18 (85%) of California’s 21 Dusky Warblers have occurred in October, this is the second latest (CBRC 2007).

RED-FLANKED BLUETAIL *Tarsiger cyanurus* (5, 2). Two Red-flanked Bluetails were banded at Southeast Farallon Island, SF: one in formative plumage 3 Nov 2019 (JRT†; 2019-162; Figure 8) and one of undetermined age 7–8 Nov 2019 (MSi†, JRT†; 2019-167). Four (80%) of the now five Red-flanked Bluetails accepted in California were on offshore islands (CBRC 2007, 2020).

WOOD THRUSH *Hylocichla mustelina* (36, 1). One was along Putah Creek, SOL, 15–19 Sep 2019 (RM§, HC§, SCH§, KS§; 2019-116). Fall records in California extend from 1 October to 25 November (CBRC 2007, 2020), making this an exceptionally early fall migrant or a summer visitor detected late in its stay.

CURVE-BILLED THRASHER *Toxostoma curvirostre* (37, 1). One in formative plumage was at Palo Verde Ecological Reserve, Blythe, RIV, 1 Jan 2019 (WR†; 2019-004). IDENTIFICATION NOT ESTABLISHED: Most committee members could not distinguish supposed Curve-billed Thrashers photographed at Palo Verde Ecological Reserve, RIV, 12 Oct 2019 (2019-142) and Joshua Tree, SBE, 27 Nov 2019 (2019-193) from other thrasher species.

WHITE WAGTAIL *Motacilla alba* (38, 1). One seen and heard calling while flying over a boat near Santa Barbara Island, SBA, 13 Oct 2019 (NL; 2019-143) was considered by the committee to be the same female in formative plumage found at San Clemente Island, LA, 15–17 Oct 2019 (NJD†, JT§; 2019-144). It is the first recorded for the Channel Islands. The bird’s bold white wing bars, blackish centers on the greater coverts, and gray crown are characteristic of the subspecies *M. a. ocularis* (see Sibley and Howell 1998), which in California tends to occur later in fall than *M. a. lugens* (CBRC 2007).

COMMON REDPOLL *Acanthis flammea* (180, 1). A female in formative plumage was at the Fish Docks, Point Reyes National Seashore (NS), MRN, 14–20 Jun 2019 (AWL†, DBar†, MDet†, DLe†, RAR†, DSS; 2019-054). Although 87% of California’s 180 accepted records come from the Modoc Plateau between 26 November and 5 March, there are now five records of late spring (3 May–27 June) migrants along the coast or on islands (CBRC 2007, 2020).
Figure 7. This Yellow-browed Warbler at Markleeville, Alpine County, photographed here 25 Oct 2019 (2019-152), was the first recorded in California.

Photo by Todd Easterla

Figure 8. This Red-flanked Bluetail at Southeast Farallon Island, San Francisco County, 3 Nov 2019 (2019-162) shows the brown greater coverts, plain tertials, and pointed rectrices indicative of formative plumage (see Leader 2009).

Photo by James R. Tietz
SNOW BUNTING *Plectrophenax nivalis* (145, 3). Three individuals were found in fall 2019: one in formative plumage at Humboldt Bay, HUM, 28 Oct 2019 (SSE†, FF†; 2019-157); a female in formative plumage at Nunes Ranch, Point Reyes NS, MRN, 8–10 Nov 2019 (MaF†, AH†, JJA†, BBo†; 2019-168); and one at Gold Bluffs Beach, HUM, 18 Nov 2019 (CNR; 2019-194). IDENTIFICATION NOT ESTABLISHED: One reported from Hayward Regional Shoreline, ALA, 11 Nov 2019 (2019-184) was insufficiently documented.

CASSIN’S SPARROW *Peucaea cassinii* (96, 41). The largest incursion of the Cassin’s Sparrow into California yet known occurred in the spring of 2019. Two prior irruptions in the Lanfair Valley in the eastern Mojave Desert involved 15 singing males in 1978 (Luther et al. 1983) and three singing males in 1993 (Erickson and Terrill 1996). All three of these events followed wet winters that resulted in unusually lush vegetation in the deserts of southern California (CBRC 2007). The first birds to arrive in 2019 were three singing males at Cabazon, RIV, 18–22 Apr 2019 (TABe†§; 2019-025), preceding the previous early date for a spring migrant by two weeks (2 May 1978, Whitewater River mouth, RIV; 1980-069; Binford 1985). At least 33 birds along Cedar Canyon Road in the Lanfair Valley, SBE, 3 May–21 Jun 2019 (TABe†‡§, AHo†§, JPi†, TGu†, DWN†, JTS§, ET†§; 2019-032; Figure 9) included 25 singing males, six females, and two juveniles. The two juveniles, a fledgling ~5.8 km east of Lanfair Road and a juvenile ~9.6 km west of Lanfair Road, established the first records of Cassin’s Sparrow breeding in California. Away from the Lanfair Valley, singing males were found on San Clemente Island, LA, 30 Apr 2019 (NJD†§, JTS†; 2019-031); in Greenwater Valley, INY, 14 May 2019 (PG†§; 2019-035); ~2.5 km north of Cima, SBE, 26 May 2019 (JC†; BT†; 2019-056); and at Camp Pendleton, SD, 19 Jun 2019 (JFGr†‡; 2019-111). The sole fall migrant was one in worn formative plumage commencing second prebasic molt on Southeast Farallon Island, SF, 4–6 Sep 2019 (JRT†; 2019-101).

FIELD SPARROW *Spizella pusilla* (19, 1). One in formative plumage was photographed at Furnace Creek Ranch, INY, 20 Dec 2019 (TL†; 2019-200). IDENTIFICATION NOT ESTABLISHED: Photographs of one reported at a residence in Bishop, INY, 18–27 Dec 2019 (2019-201) appeared to show a weakly marked White-crowned Sparrow (*Zonotrichia leucophrys*) in formative plumage that superficially resembled a Field Sparrow.

LECONTE’S SPARROW *Ammospiza leconteii* (41, 1). One in formative plumage near the Drake’s Estero trailhead at Point Reyes NS, MRN, 17 Nov 2019 (SBu†; 2019-181) extended the late date for a fall migrant in California by one day (CBRC 2007).

RUSTY BLACKBIRD *Euphagus carolinus* (74**, 7). Wintering individuals included a female at Jesse Owens Park, Los Angeles, LA, 7 Jan–4 Feb 2019 (RB†; 2019-006) and a male near the San Diego River mouth, SD, 30 Nov 2019–1 Mar 2020 (KG†, TABl†, GM, JSp, JTS†; 2019-192). A female photographed at Cypress Point in Tomales Bay State Park, MRN, 27 Sep 2019 (DLu†; 2019-127) established California’s earliest date for a fall migrant by five days (Harris 1996, CBRC 2007). Additional fall migrants were single males at Carpinteria State Beach, SBA, 5 Nov 2019 (RPO†; 2019-166); Lake Balboa, San Fernando Valley, LA (MiS†; 2019-182), and Santa Cruz Island, SBA (DD†; 2019-199), both 21 Nov 2019; and a female on San Clemente Island, LA, 26 Nov 2019 (JTS†; 2019-189). The Rusty Blackbird has averaged 4.5 accepted records per year in California over the last 15 years, leading to its removal from the review list in 2020. The CBRC reviews reports of the Rusty Blackbird from 1972 to 1974 and 2006 to 2019.

COMMON GRACKLE *Quiscalus quiscula* (101, 1). A female was at Nunes Ranch at Point Reyes NS, MRN, 13 Jun 2019 (AHi; 2019-067). All four Common Grackles
Figure 9. This Cassin’s Sparrow, photographed taking food to a nearby fledgling in the Lanfair Valley, San Bernardino County, 21 Jun 2019 (2019-032), provided the first evidence of this species breeding in California.

Photo by Thomas A. Benson

Figure 10. This male Golden-winged Warbler in formative or first alternate plumage was photographed 4 Jun 2019 at Presidio Park in San Diego, San Diego County (2019-046).

Photo by Douglas W. Aguillard
Figure 11. Most of California’s records of the Connecticut Warbler are from Southeast Farallon Island, so any mainland sighting of this skulker is a rare treat. This bird in formative plumage was photographed 9 Sep 2019 at Lila Keiser Park in Morro Bay, San Luis Obispo County (2019-110).

Photo by Janny and Steve Tillman

Figure 12. An early spring surprise was this Red-faced Warbler in formative plumage found at Wardlow Park in Long Beach, Los Angeles County, photographed here 28 Apr 2019 (2019-028).

Photo by Trish Gussler
accepted from Marin County have occurred on Point Reyes between 21 May and 16 June (CBRC 2007, 2020).

WORM-EATING WARBLER Helmitheros vermivorum (139, 4). One fall migrant was at the Goleta Monarch Grove, SBA, 5 Sep 2019 (NL†, SK†; 2019-102), and another was at the Wiyot Tribe Table Bluff Reservation, HUM, 25 Nov 2019 (RoF; 2019-186). Two individuals in formative plumage overwintered, one in Davis, YOL, 27 Oct 2019–29 Feb 2020 (ZP†, NJO†, LP†, MJR†, ZV†; 2019-155) and one at the Arcata Marsh, HUM, 22 Nov 2019–20 Mar 2020 (AL, RoF†, DK-B†; 2019-183).

GOLDEN-WINGED WARBLER Vermivora chrysoptera (83, 2). A male in formative or first alternate plumage was at Presidio Park, San Diego, SD, 4 Jun 2019 (MBr, JTS†, DWA†, EGK†, MMa†, GM, GLR; 2019-046; Figure 10). A female in formative plumage was at Dos Palmas Preserve, RIV, 11 Sep 2019 (SS†; 2019-112).

BLUE-WINGED WARBLER Vermivora cyanoptera (55, 1). An adult male in definitive basic plumage wintered at El Dorado Regional Park in Long Beach, LA, 16 Feb–26 Mar 2019 (RA†, DJB†, TABe†, TGu†, WH†, LJL†, GM, MM†, LP†, CNR, ES, JTS†; 2019-018). The only other record of a Blue-winged Warbler wintering in California is of a male in formative plumage in Ferndale, HUM, 2 Jan–7 Mar 1993 (1993-027; Erickson and Terrill 1996).

CONNECTICUT WARBLER Oporornis agilis (127, 1). Rarely recorded anywhere in California away from Southeast Farallon Island, one in formative plumage was beautifully photographed at Lila Keiser Park, Morro Bay, SLO, 9 Sep 2019 (J&ST†; 2019-110; Figure 11).

MOURNING WARBLER Geothlypis philadelphia (167, 4). Of the four Mourning Warblers documented in 2019, three were in fall and one was in spring, roughly mirroring the overall seasonal distribution of California’s records of this species. The one in spring was a male in first alternate plumage at the Silver Saddle Resort, Galileo Hill, KER, 3 Jun 2019 (MBr, JTS†; 2019-045). The three in fall comprised a male in formative plumage at Nunes Ranch, Point Reyes NS, MRN, 13 Sep 2019 (MaF†, DSS; 2019-114) and individuals in formative plumage (sex unknown) at Creekside Park, CC and ALA, 16 Sep 2019 (AMa, MJR†; 2019-115) and Huntington Central Park, Huntington Beach, ORA, 21 Sep 2019 (RSc†; 2019-125). IDENTIFICATION NOT ESTABLISHED: The report of one at Idyllwild, RIV, 12 Sep 2019 (2019-137) did not receive sufficient support from the committee.


CAPE MAY WARBLER Setophaga tigrina (51**, 3). This species occurs more frequently in fall than in spring, but the three records accepted from 2019 were all from spring. An adult male in definitive alternate plumage was on San Clemente Island, LA, 4 Jun 2019 (NJD†; 2019-047), and two single males in first alternate plumage were at Point Reyes NS, MRN, one near the Fish Docks 11 Jun 2019 (LC; 2019-052), the other at Mendoza Ranch 19 Jun 2019 (MaF†; 2019-092). IDENTIFICATION NOT ESTABLISHED: A report from Clear Lake State Park, LAK, 30 Aug 2019 (2019-210) was not accepted.

GRACE’S WARBLER Setophaga graciae (84, 5). Presumed fall migrants included individuals in formative plumage at La Jolla Colony Park, San Diego, SD, 22–23 Oct 2019 (AN†; 2019-148); at Point Loma, SD, 24 Oct 2019 (PEL; 2019-149); and at Idyllwild, RIV, 4 Nov 2019 (GA†; 2019-163). Wintering birds included a female in formative plumage at Long Beach Recreational Park, LA, 24 Jan–31 Mar 2019 (TABe†, TGu, JTS†; 2019-007); and returning adults at Villa La Jolla Park, La Jolla,
SD, 27 Sep 2019–21 Mar 2020 (ARA†; 2019-126); Encinitas, SD, 9 Nov 2019–6 Mar 2020 (TGo†, SES†; 2019-173); and Del Mar, SD, 23 Nov 2019–5 Jan 2020 (DK†; 2019-185). Individuals observed in spring and summer included a singing adult male at the Chilao Visitor Center, San Gabriel Mountains, LA, 23 Apr 2019 (RST†; 2019-026) that subsequently moved to nearby Charlton Flats, LA, 30 Apr–14 Jun 2019 (DDo†, KLG, ML†, DSt†§; 2019-037), and an adult at Crestview Rest Area, MNO, 22 Jun 2019 (BAi†; 2019-058).

**RED-FACED WARBLER** *Cardellina rubrifrons* (26, 1). One in formative plumage was at Wardlow Park, Long Beach, LA, 27–28 Apr 2019 (KM†, TABe†§, TGü†, KR†, ES†, AJS†; 2019-028; Figure 12). This record is over two weeks earlier than California’s next earliest record of a spring migrant, 13 May 1990 in Caruthers Canyon, New York Mountains, SBE (1990-096; Patten and Erickson 1994). This timing is consistent with the arrival of early spring migrants in Arizona (Martin and Barber 1995), and there are no accepted records from California in winter. IDENTIFICATION NOT ESTABLISHED: The report of one from Morris Ranch Road in Garner Valley, RIV, 1 Mar 2019 (2019-029) did not receive sufficient support from the committee.

**MISCELLANEOUS**

The long-staying Northern Gannet (*Morus bassanus*; 2012-058) first seen at Southeast Farallon Island, SF, 25 Apr 2012 (Pike et al. 2014), and the female Common Black Hawk (2005-060 et seq.) resident near Santa Rosa, SON, since 14 May 2005 (Iliff et al. 2007), were both still present through 31 Dec 2019.

**CORRIGENDA**

In the 22nd report (McCaskie and San Miguel 1999) the date span for a Violet-crowned Hummingbird (*Leucolia violiceps*) in Carlsbad, SD (1996-154), was published as 13 Nov–3 Dec 1996; the correct interval is 3 Nov–3 Dec 1996. In the Acknowledgments section of the 44th report (Benson et al. 2020), we regrettably misspelled the name of Steve N. G. Howell. Also in the last report, the statement in the Bulwer’s/Jouanin’s Petrel (*Bulweria bulwerii/fallax*) account that Bulwer’s from the western Pacific approach the larger Jouanin’s in bill size was a misinterpretation of a communication from S. N. G. Howell. Nevertheless, the question of variation in size within Bulwer’s Petrel, first addressed by Mathews and Iredale (1915), needs a quantitative analysis.

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THE 45TH CALIFORNIA BIRD RECORDS COMMITTEE REPORT

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SAGEBRUSH SOUNDSCAPES AND THE EFFECTS OF GAS-FIELD SOUNDS ON GREATER SAGE-GROUSE

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ABSTRACT: Greater Sage-Grouse (Centrocercus urophasianus) use elaborate acoustic and visual displays to attract and select mates, and females and chicks depend on acoustic communication during brood rearing. A potential threat to the grouse is sounds associated with human activity. During April, 2013–2020, we collected 17,825 hours of acoustic data in three different acoustic situations in the sagebrush of Wyoming: rural, undeveloped areas (6), at Greater Sage-Grouse leks in a natural-gas field (20), and near active machinery in that gas field (17). The average existing sound levels in undeveloped sagebrush areas were $L_{A_{eq}} = 26$ dB and $L_{A_{50}} = 20$ dB, and the average background sound level was $L_{A_{90}} = 14$ dB. These values are lower than previously reported, due in part to our use of more sensitive equipment as well as addressing the influence of the instruments’ electronic self-noise. $L_{A_{eq}}$ and $L_{A_{50}}$ at leks in the gas field ranged from 25.5 to 33.7 dB and 20.5 to 31.3 dB, respectively, depending on the distance, number, and type of nearby activities. Sound levels at leks were correlated with trends in the number of grouse using the lek: the higher the sound level, the greater the likelihood of a decline. Thresholds above which declines occurred were $L_{A_{eq}} = 31$ dB and $L_{A_{50}} = 26$ dB. Leks with $L_{A_{eq}} > 31$ dB and $L_{A_{50}} > 26$ dB, 100% and 91%, respectively, had declining trends. Our findings suggest that the current policy of limiting sound levels at leks to $L_{A_{50}} < 10$ dB (or $L_{A_{eq}} < 15$ dB) over the background sound level is appropriate, if an accurate background level is used.

Populations of Greater Sage-Grouse (Centrocercus urophasianus) have declined dramatically because of habitat loss, degradation, and fragmentation resulting from human activity such as fuel extraction, urban development, traffic on roads, and installation of power lines. Further adverse factors are improper management of grazing, altered fire regimes, and invasion by non-native annual plants (Connelly et al. 2004, Aldridge et al. 2008, Naugle et al. 2011, Boyd et al. 2014, Coates et al. 2016). Several studies have suggested that disturbance from noise pollution may be one causal mechanism by which human activities depress sage-grouse populations, on the basis of observations that attendance at leks in noisier areas is often lower (Rogers 1964, Braun et al. 2002, Connelly et al. 2004, Holloran 2005, Holloran and Anderson 2005, Connelly et al. 2011). Studies confirmed this effect experimentally by introducing sounds recorded at operating natural-gas fields to otherwise undisturbed leks and finding immediate and sustained declines in lek attendance by comparison to paired control leks, as well as increased stress hormones and altered behaviors (Blickley and Patricelli 2012, Blickley et al. 2012). However, because of the limited number of leks at which the sounds were played experimentally, this approach was not able to determine the
threshold level at which sounds from natural-gas development affect the sage-grouse. Previous research on songbirds suggests the possibility of a nonlinear threshold response to anthropogenic sound, with effects beginning when the sound is 10 dB or more above background levels (WGFD 2003, Dooling and Popper 2007). Based on this research, current management strategies for Greater Sage-Grouse in many states limit anthropogenic sound at known leks to 10 dBA above background levels. Nevertheless, we do not yet know if this threshold is appropriate for the Greater Sage-Grouse, and there is little agreement on the measures of sound appropriate for use in calculations of allowable levels (Blickley and Patricelli 2012, Patricelli et al. 2013).

The Bureau of Land Management's Record of Decision (BLM 2008) for the development of a gas field in the Pinedale Anticline project area south of Pinedale, Wyoming, included a matrix that listed species of concern and factors needing monitoring for them. The Greater Sage-Grouse was one of those species, and noise was identified as a factor to be monitored. Therefore, we identified four primary objectives of this study. (1) To determine sound levels and sound sources during the spring lekking period in rural sagebrush (Artemisia sp.) habitats in Wyoming that support the grouse and are away from gas fields. (2) To determine levels of sound generated by common activities in the Pinedale gas field. (3) To measure sound levels at Greater Sage-Grouse leks in this gas field. And (4) to assess the relationships between sound levels at leks in and near the gas field and changes in counts of male sage-grouse at those leks.

State agencies commonly use counts of males at sage-grouse leks as a measure of local abundance and trends in attendance over time (Reese and Bowyer 2007, Doherty et al. 2010). We examined the relationship between lek-attendance trends and sound levels at leks, and we make specific recommendations for the measurement, assessment, and management of anthropogenic sounds relative to the Greater Sage-Grouse.

METHODS
Study Area

In 2000, the state of Wyoming established local working groups to develop and implement conservation plans in eight regions for the benefit of the Greater Sage-Grouse and its habitats (WGFD 2003; Figure 1). These areas were selected on the basis of sagebrush habitat and known populations of the grouse. We measured sound levels in four of these areas: Bates Hole/Shirley Basin, Bighorn Basin, Upper Green River Basin, and Wind River/Sweetwater River Basin. The Pinedale gas field lies in the Upper Green River Basin and has been developed extensively for gas extraction. Its elevation ranges from 2073 m to 2286 m, and precipitation averages 30 cm annually (Western Regional Climate Center, Reno, Nevada). The vegetation is primarily big sagebrush (Artemisia tridentata), native perennial grasses such as needle and thread (Hesperostipa comata), thickspike wheatgrass (Elymus lanceolatus), Indian ricegrass (Achnatherum hymenoides), and bluebunch wheatgrass (Pseudoroegneria spicata), and forbs such as hoary tansyaster (Machaeranthera canescens), buckwheat (Eriogonum spp.), fleabane (Erigeron spp.), and phlox (Phlox spp.).
Definitions

Two sound-level metrics important in this study are the background sound level and the existing sound level. The background sound level is the level in a given environment without contributions from the source of interest (ANSI 1988, 1994). The background sound level (also often referred to as “baseline” or “residual”) is calculated as the metric $L_{A90}$, the level exceeded 90% of the time, or the level beneath which the quietest 10% of measurements lie (EPA 1971, ANSI 1988, BSI 2019; the “A” in $L_{A90}$ means the level is A-weighted, see definition below). $L_{A90}$ is generally effective at discounting intermittent anthropogenic sounds; however, background sound levels are often difficult to measure accurately because of interference by such sounds. If this interference prevents accurate measurement of background sound levels, and the responsible activities cannot be shut down, the measurements should be made in a similar habitat without any such activity (ANSI 1994). $L_{A90}$, when measured in the presence of the source of interest, is not an accurate representation of background sound level, especially when human activities generate continuous or near-continuous sound. In most rural situations where anthropogenic sounds are uncommon, $L_{A90}$ is usually effective at excluding those sources. The term “background sound level” is preferred over “baseline sound level” because “baseline” is used in the environmental review process of the National
Environmental Policy Act to mean “current conditions,” and, in acoustic metrics, “current conditions” means “existing sound level.”

The metrics $L_{A50}$ and $L_{Aeq}$ are both used to describe the existing sound level, which is the level of all sounds in a given area, including all natural and non-natural sound sources (ANSI 1988, 1994). $L_{A50}$ represents the sound level exceeded 50% of the time, or the median of all measurements (EPA 1971). It is often used to quantify the existing sound level, in part, because it is not influenced by short, loud sounds (as $L_{Aeq}$ is) and is often the best representation of “typical” sound levels at a location. $L_{Aeq}$ is the “equivalent sound level,” which is a logarithmic average (i.e., on an energy basis) of sound-pressure levels over a specific interval. $L_{Aeq}$ is useful because it does include short but loud noises, which $L_{A50}$ might miss, and such events could be important to wildlife. Both metrics are often used in characterizing levels of anthropogenic sounds relative to their potential effects on wildlife (Pater et al. 2009, Barber et al. 2011, Patricelli et al. 2013), and $L_{A50}$ is specified in some management documents (in Wyoming, Gordon 2019). Both metrics are useful in describing anthropogenic sounds, and we analyzed both in relation to grouse trends.

It is important to note that both $L_{A50}$ and $L_{Aeq}$, when measured before a proposed development, are almost certainly influenced to some degree by sounds of human activities such as ranching and farming, as well as aircraft, vehicles, and railroads. Wyoming executive order 2019-3 is specific on the $L_{A50}$ metric and its use in assessing the effect of noise, stating that “sound levels at leks, due to new project noise individually or cumulatively from anthropogenic sources, should not exceed 10 decibels (dB) above baseline at the perimeter of the lek” (Gordon 2019). In other words, the $L_{A50}$ used to assess acoustic effects must be a cumulative $L_{A50}$ of all anthropogenic sounds, including both noise due to a new project plus that due to other current human activities, and not just the $L_{A50}$ of the proposed new activity.

The background sound level is an important metric because it is the background level against which anthropogenic sounds are compared and monitored after a project has begun. The existing sound level is important because it is used to measure post-development sound levels, and it is used to determine if post-development sound levels exceed levels detrimental to the species of interest. Many current grouse-management policies state that projects’ sound levels plus sounds of other human activities shall not exceed background sound levels ($L_{A90}$) by 10 dB, as measured at the perimeter of the lek (see NDOW 2018, Gordon 2019). It is critical that both variables be measured correctly and consistently, and the only way to ensure this is to establish standards for equipment, measurement protocols, analysis, and reporting (see below).

Definitions of additional terms can be found in Barber et al. (2011) and Pater et al. (1999). Audibility is the ability of animals with normal hearing, including humans, to hear a given sound. Audibility is affected by the animal’s intrinsic ability, other simultaneous interfering sounds or stimuli, and by the frequency and amplitude of the sound. Frequency weighting is used to adjust the amplitude of various parts of the frequency spectrum for specific purposes. A-weighting (dBA) is used to account for differences in the sensitivity of human hearing as a function of frequency. A-weighting de-emphasizes low
(<500 Hz) and high (>6000 Hz) frequencies while emphasizing those in between, in an effort to simulate the response of human hearing. The noise floor (instrument self-noise) is the inherent electrical noise of all components of a sound-level meter (meter, microphone, and preamplifier), and is generally considered, although not entirely accurately, as the device’s lower measurement limit. The background noise level is the total acoustical and electrical noise, from all sources in a measurement system that may interfere with the production, transmission, time averaging, measurement, or recording of an acoustical signal. “Background noise” differs from “background sound” in that background noise is typically electrical noise in the measurement system, while background sound is the sound level in a given environment without the specific sound source of interest.

The American National Standards Institute (ANSI) has established accuracy and stability standards for three types of sound-level meters, types 0, 1, and 2. For Type 1 meters, the maximum change within one hour of operation is 0.3 dB; for Type 0, 0.2 dB; for Type 2, 0.5 dB. The maximum allowable deviation varies by frequency, with lower frequencies having tighter standards, –1.0 dB to +1.5 dB at 31.5–2000 Hz for Type 1 meters. Type 0 meters allow for roughly half this deviation, and Type 2 meters allow for roughly twice this deviation. Type 1 sound-level meters are generally used for environmental studies, although for some situations, such as long-term monitoring, Type 2 standards may be adequate.

Sound-Level Measurements

To establish background sound levels, we collected “reference” data at six locations in undeveloped sagebrush, generally 0.3–1.0 m high, and >5 km from development. The reference measurements were made in April, the primary month of sage-grouse lekking in central Wyoming, of 2013 and 2014, with an average of 19 days per site. The sites were selected in conjunction with Wyoming Game and Fish Department biologists. Three were within 150 m of leks and three were not near leks. We set a goal of at least 14 days of measurement to account for natural variability in acoustic conditions and to ensure reported levels are ±3 dBA of levels typical for the season and location. We selected this interval after considering several multiyear sets of data from national parks (Iyer 2005), but the topic needs more study. At one site, Bates/Shirley Hole, the weather restricted us to 13 days of measurements.

In the Pinedale gas field, we measured sound levels at 37 locations. Of these 37, 20 were near leks and 17 were near specific gas-field operations. The 17 measurements near gas-field operations were taken in April 2013, with an average of 1.4 days per site. As the sounds generated in gas fields tend to be consistent, varying little from day to day or year to year, these sites did not require measurement over so long an interval as measurements at leks. Measurements at the 20 leks, all during April, extended over the 8-year period 2013–2020, with an average of 5.2 days per lek per year. This 8-year interval is shorter than the period over which grouse were counted at the leks (2000–2020), and this could introduce unexplained variation into our analysis. However, we chose to use the longer interval for calculation of trends rather than restricting our calculations to 2013–2020 for two reasons.
First, the gas field has been in operation essentially from 2000 onward. There were only three well pads prior to 2000, and the majority of development was from 2001 to 2005. Relatively few additional well pads have been constructed since 2006 (we were unable to obtain information on the number of pads constructed per year from the companies operating them, and relied on Google Earth imagery for these observations). Since 2005, the distance between most leks and the nearest well pad has changed little. The distance from one lek to the nearest well pad decreased by roughly 50%, and, assuming an increase of 6 dB per halving of distance (or a decrease of 6 dB per doubling of distance, see OSHA 2013), we estimated that sound levels at this lek increased by 6 dB. Second, although seven of the leks in our study area were abandoned before 2013, excluding these leks would remove from our analysis those leks that may have been most strongly affected by development. For our analysis, therefore, we assumed that sound levels at leks from 2013 to 2020 were representative of sound levels following the initial period of development (2001–2005), except at the one lek where we estimated that the sound level increased by 6 dB. Overall, annual variation in sound levels for the period 2013–2020 was small: the mean standard deviation in $L_{A50}$ was 2.2 dB (range 0.8–4.1). At the seven leks abandoned before our acoustic study began in 2013, we estimated sound levels from the averages during our first two years of measurements. At the leks abandoned during our study, we used data through the last year of occupancy.

Protocols for Measuring and Reporting Sound Levels

Because measurements of sound levels in gas fields could be used as evidence of habitat degradation, they are controversial and open to criticism. Therefore, they should be taken with ANSI Type 1 equipment rated to a sensitivity appropriate for the acoustic conditions of each study area. In this study, we used equipment with a noise floor of 14 dBA, but more sensitive sound-level meters are now available (noise floor 5.5 dBA). The protocol we developed for measuring and reporting sound levels in sagebrush is presented in detail in the Appendix, available at https://archive.westernfieldornithologists.org/archive/V52/Ambrose_Appendix/.

Instrumentation

We used Larson-Davis 831 sound-level meters with PCB 377B20 microphones, Larson-Davis PRM831 preamplifiers, and Larson-Davis EPS2106 environmental shrouds (open-cell foam windscreen and spike to discourage birds from perching on it; Larson-Davis, Provo, UT). Sound-level meters, microphones, preamplifiers, and environmental shrouds met ANSI Type 1 standards. We used Roland R05 digital recorders (Roland, Los Angeles, CA) with Wildtronics Micro Mic PIP microphones (Wildtronics, LLC, Newton Falls, OH) and 90-mm foam windscreens (GRAS Sound and Vibration, Beaverton, OR) to make continuous recordings (16-bit, MP3, 128 kilobytes per second). We placed equipment at the perimeter of leks in sagebrush >0.3 m tall to hide equipment from grouse view and to serve as a windscreen. Microphones were 0.3 m above ground, the average height of a grouse’s ear. We used a Bruel & Kjaer 4231 (Bruel & Kjaer, Norcross, GA) field calibrator to check calibration.
at the beginning and end of each measurement period. All instruments were calibrated according to the manufacturer’s standards and schedules. We synchronized all equipment components at the beginning of the measurement period by means of the GPS. Figure 2 shows a typical setup of the equipment.

In 2013 and 2014, we used Young Model 12102 anemometers (R. M. Young Co., Traverse City, MI) connected directly to the LD 831 to record wind speed once per second. Wind blowing on the foam windscreen can influence the sound levels the instruments record. The sound of wind through vegetation is natural, but the sound of wind on the foam windscreen is not. Such “pseudo noise” is usually addressed by excluding data collected when wind speeds exceed 5 m/sec. (ANSI 1994). However, at the 0.3-m height we placed our microphones and in sagebrush >0.3 m, wind speed exceeded 5 m/sec <0.02% of the time. Therefore, the influence of pseudo noise was minimal, and we did not exclude any data from analysis on this basis.

Sound-Level Metrics

Sound-level meters collected continuous 1-second dBA, dBC, and dBF, as well as unweighted one-third octave band levels, 12.5–20,000 Hz, for the entire measurement period. From these 1-second data, we computed 1-hour values for $L_{A_{eq}}, L_{A10}$, $L_{A_{50}}, L_{A90}, L_{A_{min}}$, and $L_{A_{max}}$. Then from these hourly summaries, we computed average (arithmetic mean) sound levels
for all hours of the day (00:00–24:00) as well as for the period during which sage-grouse attend leks (18:00–08:00; all times Mountain Daylight Time). The interval 18:00–08:00 is specified in several guidelines for Greater Sage-Grouse management (NDOW 2018, Gordon 2019). For sites for which we had multiple years of data, we calculated the arithmetic mean of the annual levels for a long-term average. We used the arithmetic mean of all reference sites in undeveloped sagebrush to represent overall levels of background and existing sound for Wyoming.

Influence of Sound-Level Meters’ Electrical Self-Noise

The components of sound-level meters produce inherent electrical self-noise, such as that introduced by the microphone, preamplifier, and power supply. When actual sound levels are within 10 dB of a meter’s electrical self-noise (noise floor), they are lower than the level the meter reports and should be corrected. The sound-pressure level that a meter displays is actually the addition of two electrical signals: the instrument’s electrical self-noise plus the actual environmental sound. Added logarithmically, two sound levels of equal magnitude yield a reading 3 dB greater than the sound level from one of these sources. For example, if the electrical self-noise of a sound-level meter is 13.0 dBA, and the actual sound level is 13.0 dBA, the meter reads 16.0 dBA. When the electrical self-noise of a sound-level meter is well understood, its readings can be corrected for noise-floor influence by decibel subtraction.

While sound levels corrected for self-noise may not meet ANSI Type I standards (specifically, corrected sound levels may not be ±1 dB of actual levels), corrected levels are generally more accurate than uncorrected levels and should be reported when levels are within 10 dB of the instrument’s noise floor. Ideally, sound levels should be measured with meters sensitive enough to preclude the need for noise-floor correction; however, if low-noise meters are not available, readings should be corrected for the noise floor. The meters we used in this study had noise floors of 13–15 dBA. At our six reference measurement sites, most values of $L_{A50}$ (69%) and $L_{A90}$ (83%) were within 10 dB of the instruments’ noise floor. Hence most of the levels reported by the sound-level meter were higher than actual levels, and thus the need for noise-floor correction.

In 2014, at one of our reference sites, we tested this correction method by comparing 5 days of data collected with a standard ½” microphone (noise floor 13.0 dBA) with data collected simultaneously by a very low-noise GRAS 1” microphone (noise floor 0 dBA). Using the known noise floor and decibel subtraction, we corrected levels of the ½” microphone and compared the three data sets (1-inch microphone, $L_{A90} = 14.1$ dB; ½” microphone uncorrected, $L_{A90} = 18.7$ dB; and ½” microphone corrected, $L_{A90} = 14.4$ dB). Given this close agreement between actual and corrected levels, we corrected values for all metrics. We did not correct values to below 0 dBA, although A-weighted sound levels can be less than 0 dBA, the lower limit of human hearing.

Measurements of Sound Levels in Active Gas Fields

We measured the sounds generated by common gas-field activities at 17 locations in the Pinedale gas field, and at four of these we deployed mul-
multiple sound-level meters to assess attenuation rates. We followed Mueller’s (2002) recommendation to place the meters two source widths away from the source. At a well pad, say 150 m across, sound is typically generated by several simultaneous activities, such as drilling, generators, and vehicles. In this situation, we placed the sound-level meter 300 m from the drill pad (150 m × 2). But because of interference from other nearby sound sources this was not always possible. At four locations, we placed two sound-level meters at different distances from sound sources, and, whenever possible, we doubled the distance between source and each meter to check attenuation rates. Although we often had to measure gas-field noise at different distances, to offer a consistent comparison of sound levels at the same distance, we present values recalculated to represent sound levels at 100 m. At the four locations with two sound-level meters, with the second at double the distance from the source as the first, we found an average attenuation rate of 6.6 dB (range 6–8 dB) per doubling of distance, very close to the expected value (OSHA 2013). We used a Leica LRF 1200 laser rangefinder to measure the distance from sound source to sound-level meter. To determine the distance from a lek to the nearest active well pad, we measured the distance from the centers of each with Google Earth, which offers imagery for several years during our study period.

Audibility and Digital Recordings

The sound level in decibels alone does not allow identification of all sources of sounds. In acoustic studies, it is useful to know the source of common sounds, both natural and non-natural, and the percent time that such sounds are audible (Francis and Barber 2013). For this we listened to a subsample of the audio recorded during the study. At six sites outside the gas field, we listened to 10 seconds of recording every 4 minutes over 2 days. This scheme captures most non-natural transient sounds such as the passage of aircraft and vehicles (Ambrose unpubl. data). The 10-second/4 minute scheme resulted in a 1-hour recording for each day sampled. All sound sources were logged into a spreadsheet from which we computed audibility. We selected two days randomly at each site, excluding days of inclement weather or strong wind, in accordance with the lek-count protocol. We did not assess audibility at sites in the gas field because the sounds it produces were audible 100% of the time at most locations. At occupied leks outside the gas field, we logged the sounds of displaying grouse during each 10-second segment to ascertain the daily cycle of display.

We randomly selected the days for audibility analysis at each lek for one weekday and one weekend day, with the primary purpose of documenting sources of natural and non-natural sounds, and the percent time that each was audible. We did not select these days with respect to grouse displays or to the phase of the moon. We assessed grouse display sounds relative to the phase of the moon by defining phases from first quarter to third quarter as nights with moonlight and phases from third quarter to first quarter as those without. We compared the differences in the percent of samples with grouse sounds relative to nights with and without moonlight with a paired t-test.
Frequency Weighting for the Greater Sage-Grouse

When the effects of anthropogenic sounds on wildlife are assessed, sound levels should be weighted by frequency to match the hearing ability of the target animal as closely as possible (Pater et al. 2009). The hearing abilities of most animals are not well understood, but the hearing of many species of birds is most sensitive in the same range as human hearing is most sensitive, 1–6 kHz (Fay 1988, Beason 2004). Dooling and Popper (2007) suggested that, in the absence of audiograms for the target species, the hearing of most birds is best approximated by A-weighting, and A-weighted levels are likely the best predictor of disturbance from anthropogenic sounds. Therefore, we used A-weighted sound levels in our analysis.

Counts and Trends of Male Greater Sage-Grouse at Leks

Personnel from the Wyoming Game and Fish Department (WGFD) and Bureau of Land Management (BLM) counted male sage-grouse at all known leks in \( n = 20 \) or near \( n = 3 \) the Pinedale gas field annually, following protocols outlined by WGFD’s Sage-Grouse Technical Committee (WGFD 2003). Results of counts at satellite leks (defined as a lek with <15 breeding males within 500 m of a larger lek nearby and assumed to represent the same breeding population) were folded into those of the larger lek. We omitted data from one recently discovered lek in the gas field that had only four years of data, so our assessments were based on 22 leks. The reference leks were similar in habitat and topography to those in the gas field but 7–11 km away from it. Some leks have been counted since 1990, but because most development of the gas field and consistent lek counts started in the early 2000s, we calculated trends on the basis of count data from 2000 to 2020. We considered a lek no longer occupied if no grouse were observed at it during the last two count years, consistent with Western Association of Fish and Wildlife Agency standards (WAFWA 2015). We considered a lek to be occupied if grouse were observed in two or more consecutive years.

We used generalized linear regression with a negative binomial distribution (negative binomial regression) to assess trends. Negative binomial regression is commonly used when the dependent variable is a count and zero or missing values are common (Harju et al. 2010). We used the negative binomial regression coefficient and associated \( P \) value to establish if lek counts were stable (no significant change, \( P > 0.05 \)) or increasing or decreasing significantly (\( P < 0.05 \)).

Statistical Analysis

When a new gas well is drilled, its effects on the grouse may include habitat loss, new access roads, vehicle traffic, human activity, noise, light pollution, visible structures, dust, and an increase of predators. Many of these factors are strongly correlated, and we could not single out the contribution of any one variable. For analysis of the relationship between trends in grouse numbers and sound levels at leks, we used Pearson's product moment correlation (\( r \)). To identify sound-level thresholds above which trends tend to decline, we used two-segment piecewise regression (Toms and Lesperance 2003, Ficetola and Denoël 2009). For each piecewise regression, we also calculated the \( R^2 \),
which quantifies how much variability in the response variable is explained by the model, as well as the standard error around each threshold. For these analyses, we used SigmaStat (San Jose, CA) and NCSS 12 (Kaysville, UT).

RESULTS


We collected 2805 hours of acoustic data at the six reference sites. Mean sound levels for all sites and all hours combined, corrected for the noise floor, were $L_{Aeq} = 26 \text{ dB (range 22–30 dB)}$, $L_{A10} = 27 \text{ dB (range 23–32 dB)}$, $L_{A50} = 19 \text{ dB (range 15–24 dB)}$, and $L_{A90} = 14 \text{ dB (range 10–19 dB)}$. Mean sound levels, corrected for the noise floor, from 18:00 to 08:00, were $L_{Aeq} = 24 \text{ dB (range 18–28 dB)}$, $L_{A10} = 16 \text{ dB (range 11–21 dB)}$, $L_{A50} = 11 \text{ dB (range 7–16 dB)}$, and $L_{A90} = 8 \text{ dB (range 4–12 dB)}$.

Sound Sources in Sagebrush, 2013–2014

In rural, undeveloped Wyoming, natural sounds were audible on average 71% of the time. The most common were wind through vegetation (49%), birds (35%), insects (8%), rain (2%), and mammals (1%). Non-natural sound sources were less common, audible, on average, 36% of the time. The most common non-natural sounds were jet aircraft (10%), propeller aircraft (4%), vehicles (11%), and unidentified motor sounds (12%). Sounds were not audible 13% of the time.

We analyzed 792 hours of digital recordings at 17 leks for display sounds of Greater Sage-Grouse. These included not only four leks in our study area in Wyoming but also seven observed in similar studies in northern Nevada and six in northern Utah (Ambrose unpubl. data). For all hours (00:00–24:00), 15.0% of the samples had grouse sounds; for daytime hours (08:00–18:00), 2.7% of the samples had grouse sounds; and for nighttime hours (18:00–08:00), 23.8% of the samples had grouse sounds. Of all display sounds, 92.5% were recorded from 18:00 to 08:00, and 45.3% were recorded from 04:00 to 08:00. During this 04:00–08:00 period, the amplitude of display sounds was considerably higher than at other hours (see Figure 3), suggesting this interval was the most important for display. The 06:00 hour had the highest percentage of display sounds, 50.9%. Grouse displayed during all hours of the night, regardless of the phase of the moon. The percent of samples with display sounds during nights with the moon >50% full did not differ significantly from nights with the moon <50% full (paired $t$-test, $t = 2.26$, d.f. = 9, $P = 0.62$). In Figure 3, the elevated sound levels from 04:00 to 08:00 were due to grouse display sounds.

Sound Levels of Common Gas-Field Activities, 2013

In April 2013, we collected 533 hours of acoustic data on common gas-field activities at 17 locations in the Pinedale gas field; at four of these, we made simultaneous measurements at different distances. We were not able to measure all activities at the same distance, so we modeled $L_{A50}$ from all sources at a distance of 100 m to facilitate comparison. The sound of an active drill rig was the loudest, $L_{A50} = 62 \text{ dB at 100 m}$, followed by an injection well facility,
We collected 17,407 hours of acoustic data at 20 leks in the Pinedale gas field. Sound levels varied with distance from the nearest gas-field activity and the type of activity. Sound levels at leks and the distance to the nearest development were negatively correlated: the closer the lek to a pad, the higher the sound level ($L_{A50}: r = -0.779, P < 0.001$). Piecewise regression suggested a breakpoint in the relationship with $L_{A50}$ at 3149 ± 572 m (mean ± S.E, $F = 21.9$, d.f. = 3, $P < 0.001$, $R^2 = 0.785$) and a breakpoint in the relationship with $L_{Aeq}$ at 3117 ± 619 m (mean ± S.E, $F = 16.3$, d.f. = 3, $P < 0.001$, $R^2 = 0.731$). At distances greater than these, sound levels appeared uninfluenced by most gas-field activities (Figure 4), although some loud, relatively short-term activities, such as drilling, could extend this distance. Sound levels at leks were largely dependent on the distance to nearest development or gas-field activity, with sound levels at leks >3200 m from development generally not influenced by anthropogenic sources. For this reason, we distinguish sound-level metrics for leks <3200 m and >3200 m from development (Table 1). Mean sound levels at five leks >3200 m from the nearest pad were $L_{Aeq} = 25.2$ dB, $L_{A50} = 54$ dB at 100 m. Sound levels of other gas-field activities were a drill rig being disassembled, $L_{A50} = 54$ dB at 100 m, compressor stations, $L_{A50} = 47–54$ dB at 100 m, and a central gathering facility with a generator, $L_{A50} = 45$ dB at 100 m. At the majority of pads, with active well pumps, sound levels ranged from 32 to 49 dB at 100 m, depending on the number of wells on the pad.

**Sound Levels at Leks in the Pinedale Gas Field, 2013–2020**

Figure 3. Percentage of hours (gray bars) with Greater Sage-Grouse display sounds at 17 leks (4 in Wyoming, 7 in Nevada, 6 in Utah; 1180 total hours), and mean hourly values of $L_{A50}$ (♢) and $L_{Aeq}$ (○), April 2014–2019. Elevated sound levels from 04:00 to 07:00 are due to grouse display sounds.

$L_{A50} = 56$ dB at 100 m. Sound levels of other gas-field activities were a drill rig being disassembled, $L_{A50} = 54$ dB at 100 m, compressor stations, $L_{A50} = 47–54$ dB at 100 m, and a central gathering facility with a generator, $L_{A50} = 45$ dB at 100 m. At the majority of pads, with active well pumps, sound levels ranged from 32 to 49 dB at 100 m, depending on the number of wells on the pad.
19.4 dB, and $L_{A90} = 14.7$ dB, similar to background sound levels as measured at the six reference sites ($L_{Aeq} = 25.8$ dB, $L_{A50} = 19.1$ dB, and $L_{A90} = 14.0$ dB). Mean sound levels at 17 leks <3200 m from the nearest pad were $L_{Aeq} = 30.1$ dB, $L_{A50} = 26.7$ dB, and $L_{A90} = 23.3$ dB. At leks far from gas-field activity, sound levels followed the typical daily pattern of being lowest during the evening and early morning and highest during daylight, owing primarily to higher wind speeds during daytime hours (Figure 5, PAPA103). At leks close to gas-field activity, sound levels varied little regardless of the time of day (Figure 5, PAPA001), as active wells run continuously. Over the eight years of measurements, sound levels at leks in the Pinedale gas field varied little (SD = 2.4 dB, range 0.8–4.4 dB) unless activities at nearby pads changed substantially.

**Table 1** Sound Levels\(^a\) at Greater Sage-Grouse Leks <3200 m and >3200 m from Gas-Field Operations in Wyoming

<table>
<thead>
<tr>
<th>Distance</th>
<th>Metric</th>
<th>$L_{Aeq}$</th>
<th>$L_{A10}$</th>
<th>$L_{A50}$</th>
<th>$L_{A90}$</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;3200 m ($n = 17$)</td>
<td>Mean</td>
<td>30.1</td>
<td>31.9</td>
<td>26.7</td>
<td>23.3</td>
</tr>
<tr>
<td></td>
<td>Min</td>
<td>24.1</td>
<td>25.2</td>
<td>24.1</td>
<td>14.2</td>
</tr>
<tr>
<td></td>
<td>Max</td>
<td>33.9</td>
<td>35.5</td>
<td>31.9</td>
<td>29.4</td>
</tr>
<tr>
<td>&gt;3200 m ($n = 5$)</td>
<td>Mean</td>
<td>25.2</td>
<td>26.7</td>
<td>19.4</td>
<td>14.7</td>
</tr>
<tr>
<td></td>
<td>Min</td>
<td>21.6</td>
<td>22.8</td>
<td>21.6</td>
<td>10.1</td>
</tr>
<tr>
<td></td>
<td>Max</td>
<td>28.0</td>
<td>29.0</td>
<td>23.0</td>
<td>19.1</td>
</tr>
</tbody>
</table>

\(^a\)Corrected for noise floor influence.
It is difficult to track trends in cyclic species such as sage-grouse unless count data are available for one or more cycles. In the Pinedale area, sage-grouse numbers appear to be roughly on a 10-year cycle (this study), and the difference between high counts and low counts in the cycle can be large, with high-year counts being often more than double low-year counts. With count data for two cycles (21 years), we were able to assess the longer-term trend. At the 22 leks analyzed, seven were occupied during all 21 years of lek counts (2000–2020), eight were abandoned, six were new or newly discovered, and one became occupied, was used for eight years, and then abandoned. At the 19 leks in the gas field with long-term counts, the trend at 12 was declining, at the other seven, stable or increasing (Table 2). At the three reference leks outside the gas field, grouse numbers were stable. While the number of leks occupied annually in the gas field remained relatively stable through our study (mean = 12.8, range 11–15), the total number of male grouse counted annually at these leks declined significantly (Poisson regression, $b = -0.02$, $\chi^2 = 161.5$, $P < 0.001$).

The mean sound levels at the 10 leks where the numbers of grouse were stable or increasing were $L_{A50} = 21.9$ dB and $L_{Aeq} = 26.8$ dB, whereas at the 12 leks where grouse were declining, mean levels were $L_{A50} = 27.8$ dB and $L_{Aeq} = 30.7$ dB (Figure 6). In both metrics, the differences between stable and declining were significant. For $L_{A50}$: $t = -3.781$, df = 20, $P = 0.001$; for $L_{Aeq}$: $t = -3.391$, df = 20, $P = 0.003$.

Influence of Gas-Field Sounds on Trends

Our piecewise regression analyses suggest there are thresholds of $L_{Aeq}$ and $L_{A50}$ above which the numbers of Greater Sage-Grouse displaying at a lek tend to decline. For $L_{Aeq}$ the threshold was $31.1 \pm 0.6$ dB (mean ± S.E, $F = 8.33$, $\chi^2 = 161.5$, $P < 0.001$).
d.f. = 3, \( P = 0.001 \), \( R^2 = 0.581 \)); for \( L_{A50} \), \( 26.2 \pm 1.2 \) dB (\( F = 7.08 \), d.f. = 3, \( P = 0.002 \), \( R^2 = 0.541 \)) (Figure 7).

At the 11 leks with \( L_{A50} > 26 \) dB, the trend was declining at 10 and stable or increasing at only one (Figure 8). At the 11 leks with \( L_{A50} < 26 \) dB, the trend at two was declining and at nine was stable or increasing. The mean trend at leks with \( L_{A50} < 26 \) dB was \( -0.032 \), essentially stable (\( P = 0.204 \)), whereas at leks with \( L_{A50} > 26 \) dB the mean trend was \( -0.242 \), a significant decline (\( P < 0.001 \)). At all seven of the leks where \( L_{Aeq} > 31 \) dB, the trend was declining. Of the other 15 leks where \( L_{Aeq} < 31 \) dB, the trend was flat at 10 and declining at five. There was considerable overlap in trends by both measures: the mean for leks where \( L_{A50} < 26 \) dB was \( -0.032 \) (range \( -0.249 \) to \( 0.059 \)); for those where \( L_{A50} > 26 \) it was \( -0.242 \) (range \( -0.707 \) to \( 0.113 \)). At the 15 leks where \( L_{Aeq} < 31 \) dB, the mean trend was \( -0.050 \) (range \( -0.249 \) to \( 0.131 \)); at the 7 where \( L_{Aeq} > 31 \) dB, it was \( -0.324 \) (range \( -0.707 \) to \( -0.051 \)) (Figure 8).

---

### Table 2  Trends in Greater Sage-Grouse Numbers and Sound Levels at 26 Locations in Wyoming, 2013–2020

<table>
<thead>
<tr>
<th>Lek/site(^a)</th>
<th>Trend(^b)</th>
<th>( P )</th>
<th>Stable or declining?</th>
<th>( L_{Aeq} )</th>
<th>( L_{A50} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>PAPA001</td>
<td>-0.479</td>
<td>0.00</td>
<td>D</td>
<td>34.1</td>
<td>31.6</td>
</tr>
<tr>
<td>PAPA002</td>
<td>-0.111</td>
<td>0.05</td>
<td>D</td>
<td>31.3</td>
<td>27.9</td>
</tr>
<tr>
<td>PAPA003</td>
<td>0.158</td>
<td>0.00</td>
<td>S</td>
<td>30.6</td>
<td>27.7</td>
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<tr>
<td>PAPA004</td>
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<td>23.7</td>
</tr>
<tr>
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<td>27.4</td>
</tr>
<tr>
<td>PAPA006</td>
<td>-0.070</td>
<td>0.00</td>
<td>D</td>
<td>32.2</td>
<td>29.5</td>
</tr>
<tr>
<td>PAPA007</td>
<td>-0.049</td>
<td>0.00</td>
<td>D</td>
<td>32.2</td>
<td>29.5</td>
</tr>
<tr>
<td>PAPA008</td>
<td>-0.192</td>
<td>0.00</td>
<td>D</td>
<td>32.7</td>
<td>30.5</td>
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<tr>
<td>PAPA009</td>
<td>-0.078</td>
<td>0.00</td>
<td>D</td>
<td>32.3</td>
<td>29.6</td>
</tr>
<tr>
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<td>0.00</td>
<td>D</td>
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<td>0.05</td>
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<td>32.4</td>
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<td>D</td>
<td>32.4</td>
<td>28.9</td>
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<td>0.00</td>
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<td>25.9</td>
<td>20.5</td>
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<td>17.4</td>
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<tr>
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<td>NA</td>
<td>NA</td>
<td>30.1</td>
<td>24.1</td>
</tr>
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<td>REF107</td>
<td>NA</td>
<td>NA</td>
<td>NA</td>
<td>28.1</td>
<td>20.6</td>
</tr>
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</table>

\(^a\)One lek with only 4 years of count data not included; 3 reference sites not near leks.

\(^b\)Negative binomial.
DISCUSSION

Sound Levels in Sagebrush Habitats in Wyoming

Sound levels in undeveloped sagebrush of Wyoming in April were generally low, particularly during the night and early morning when wind speeds were low. Even including low levels of anthropogenic sound, we found these levels were much lower than previously reported (BLM, Pinedale Anticline Project Office, unpubl. reports). This was due to our use of more sensitive equipment and addressing the instrument’s electrical self-noise. Daytime

![Box plots of sound levels (A, $L_{A50}$; B, $L_{Aeq}$) at Greater Sage-Grouse leks in and near the Pinedale gas field where numbers of grouse are stable ($n = 10$) and declining ($n = 12$). Box represents 75% of data; thick horizontal line; mean, thin horizontal line, median; dots, minimum and maximum levels; whisker marks, 5th and 95th percentiles.](image)

![Trends (negative binomial) in numbers of Greater Sage-Grouse at 22 leks versus sound levels quantified as $L_{A50}$ (A) and $L_{Aeq}$ (B). Piecewise regression thresholds: $L_{A50} = 26.2$ dB; $L_{Aeq} = 31.1$ dB.](image)
levels were generally 5–10 dBA higher than nighttime levels, primarily because of the sound of wind through vegetation. Anthropogenic sounds, mostly distant aircraft and vehicles, were audible on average 34% of the time, all hours pooled, with little difference between daytime (36%) and nighttime (32%). During some hours, anthropogenic sounds were audible >50% of the time, likely influencing $L_{A50}$ during these hours.

Influence of Grouse Display Sounds on Measured Sound Levels

Gordon (2019) and WGFD (2019) recommended measuring sound levels at the perimeter of the lek, with the goal of assessing increases due to anthropogenic sources. At leks with large numbers of displaying grouse, grouse displays themselves elevate sound levels (Figure 3). One potential solution is to measure sound levels away from the lek but at an equal distance to the nearest anthropogenic source. The appropriate distance from a lek depends on the number of grouse at the lek, as sound levels increase with the number of displaying birds. On the basis of our measurements, we recommend a distance $\geq 250$ m from the perimeter of the lek. Although at this distance grouse are usually audible, the influence of their displays on sound levels is generally minimal. A drawback of this approach is reducing the value of recordings for determining the presence of grouse at a lek. If the primary goal is to quantify sound levels at leks relative to anthropogenic sources, measurements should be made $>250$ m from the perimeter of the lek. If the primary goal is to study grouse biology, measurements should be made at the perimeter of the lek.

Relationship of Anthropogenic Sounds to Lek-Count Trends

Anthropogenic sounds have a significant effect on trends in counts at Greater Sage-Grouse leks above $L_{A50} = 26$ dB and $L_{Aeq} = 31$ dB. Descriptive statistics confirm these relationships: at 11 leks where $L_{A50} > 26$ dB, the trend was declining at all but one; of the 7 leks where $L_{Aeq} > 31$ dB, all were in decline.
The relationships between trends and sound levels were slightly stronger for all 24 hours of the day combined than for hours of lekking (18:00–08:00) only:

- $L_{A50}$, all hours: $r = -0.504$, $P = 0.017$ versus 18:00–08:00: $r = -0.488$, $P = 0.021$;
- $L_{Aeq}$, all hours: $r = -0.509$, $P = 0.016$ versus 18:00–08:00: $r = -0.479$, $P = 0.024$.

While our analysis demonstrated a significant relationship between trends in grouse numbers and sound levels, we are not suggesting that other aspects of gas-field activity do not contribute to declines in lek attendance as well. For example, we measured the distance from the lek to the nearest well pad in Google Earth and also the percentage of the area within 3 km of the lek that had been disturbed, using imagery from the University of Wyoming's Geographic Information Science Center. Both the distance to nearest well pad and percent area disturbed within 3 km were strongly correlated with trend at the lek ($R^2 = 0.485$, $P = 0.007$ and $R^2 = 0.772$, $P < 0.001$, respectively). The threshold for an effect of well-pad distance was 1560 m and for percent area disturbed was 4.4%. Well-pad distance and percent area disturbed were also correlated with $L_{Aeq}$ ($R^2 = 0.731$, $P < 0.001$ and $R^2 = 0.799$, $P < 0.001$, respectively) and $L_{A50}$ ($R^2 = 0.785$, $P < 0.001$ and $R^2 = 0.841$, $P < 0.001$, respectively). With this high degree of multicollinearity, we could not assess the relative importance of any these variables relative to trend. For several other possible contributing variables, such as number of predators, light pollution, dust, vehicle numbers, and level of activity at each pad, we have no data and could not compare to trend. Some of these factors are likely correlated with sound and may contribute to the relationship between sound levels and declines in lek attendance. Other factors, such as habitat loss to wildfire or conversion to cheat grass ($Bromus tectorum$), have no relationship to sound but have documented adverse effects on Greater Sage-Grouse (Ielmini et al. 2015). In our analyses of well-pad distance and percent area disturbed, we found declining trends at all leks <1560 m from a well pad and 85% of leks with >4.4% area within 3 km of the lek disturbed.

Our analysis implicates anthropogenic sound as a major negative influence on grouse in the Pinedale gas field, and examination of other types of development, even different types of gas-field development, would likely find factors other than anthropogenic sounds detrimental to Greater Sage-Grouse. It is difficult or impossible to assess the relative importance of any factor because of the high degree of multicollinearity among the variables. The ideal way to isolate the effects of various types of disturbance on population declines is to introduce each possible source experimentally in a controlled way. While it is not feasible to experimentally introduce, for example, a large-scale increase in the percentage of area disturbed without also introducing other factors, it is feasible to experimentally introduce sound. In an experimental study of introduced road sounds in Idaho, McClure et al. (2013) observed declines in several species of songbirds and almost complete avoidance by others. Blickley et al. (2012) broadcast gas-field sounds (recorded in the Pinedale field) at otherwise undisturbed leks at another site in Wyoming and found declines in attendance, altered behaviors, and evidence of increased stress in grouse at experimental leks in comparison to control leks. Ware et al. (2015) introduced traffic noise in a roadless area during autumn migration and found that 31% of the bird community avoided the experimental area. Taken
together with ours, these studies support the hypothesis that noise alone can depress sage-grouse populations.

Thresholds of Acoustic Effects on Other Species

Barber et al. (2011) reviewed four studies that assessed traffic volumes and distances at which various species are adversely affected by anthropogenic sounds. These four studies did not measure sound levels, but from their data on traffic volume and distance, Barber et al. (2011) calculated sound levels above which the animals were affected. The threshold for frogs in Ontario was $L_{Aeq} = 43.6$ dB, for grassland birds in Massachusetts $L_{Aeq} = 38.3$ dB, for woodland birds in the Netherlands $L_{Aeq} = 42–52$ dB, and for grassland birds in the Netherlands $L_{Aeq} = 47$ dB. These studies did not investigate background sound levels ($L_{A90}$), so how these threshold levels compare to background levels is not known. Barber et al. (2011) did not calculate $L_{Aeq}$ over 24 hours by averaging hourly levels as we did but used a 24-hour $L_{Aeq}$. Our approach followed Plotkin (2001), who recommended the hourly approach to account for hour-to-hour variations. In addition, the activities of many species of wildlife are associated with specific hours of the day, favoring analysis of sound levels on an hourly basis. If we calculated $L_{Aeq}$ as did Barber et al. (2011), the threshold level for the Greater Sage-Grouse would be $L_{Aeq} = 36.0$ dB, making it one of the species most sensitive in comparison to the others studied.

Management of the Greater Sage-Grouse

Our measurements and analysis suggest that plans for monitoring and management of the Greater Sage-Grouse should consider sound levels directly and not rely solely on other measures such as distance or number of wells per unit area. Sound levels at leks were closely linked to the distance to the nearest pad, the number of pads near a lek, and percent area disturbed near a lek. However, even if the effects of distance, number of pads, or habitat loss are each kept low, a cumulative increase in sound levels at a lek could lead to a decline. Pater et al. (1999) argued that while distance from a sound source is often used as a surrogate for sound disturbance, the use of properly measured sound levels facilitates a more robust analysis of potential adverse effects. Barber et al. (2011) and McKenna et al. (2016) also stressed the need for and importance of properly measured sound levels in understanding and managing acoustic effects on wildlife. Comprehensive management plans should set specific guidelines (or limits) for sound levels at leks in gas fields, in addition to other factors commonly used, such as distance to wells, number of pads, and habitat loss.

Our measurements and analyses also suggest that the current approach to managing the effects of anthropogenic sounds on the Greater Sage-Grouse in Wyoming’s executive order 2019-3 (Gordon 2019), which limits anthropogenic sounds (both from a new project individually or cumulatively from other anthropogenic sources) to no more than 10 dB over background sound levels, is appropriate. We found that the WGFD’s (2019) protocol is the proper approach to measuring and reporting sound levels at Greater Sage-Grouse leks. We suggest, however, that the location of measurement with respect to a lek should be reconsidered, perhaps moved farther from the lek to re-
duce the influence of grouse display sounds on overall sound levels (while maintaining an equal distance to the nearest anthropogenic sound source). A standardized approach to data collection and reporting is necessary for the accurate acoustic data essential for managing the effects of noise on the Greater Sage-Grouse.

We found that relationships between sound levels and trends in lek attendance were nearly identical whether the comparison was based on all 24 hours of the day or restricted to the hours of lekking (18:00–08:00). This result suggests that female–chick communication may be as important to the grouse as communication between displaying males and females. Therefore, as a basis for background sound levels, the $L_{A90}$ for all 24 hours of the day may be more appropriate than $L_{A90}$ based on the hours of lekking. Wyoming's executive order 2019-3 (Gordon 2019) defines the interval 1 March–15 May as the “breeding season,” and this is when most measurements have been made. However, the full breeding season, from displays at leks through the time when young are independent, is likely equally important. Furthermore, anthropogenic sounds, by inhibiting predator avoidance and flock communication, may have a detrimental effect on sage-grouse year round. We recommend that plans for managing the Greater Sage-Grouse include acoustic protections for the entire breeding season and all hours of the day, and that study of potential effects during the nonbreeding season be initiated.

Influence of Instrument Noise Floor

In acoustic studies of the Greater Sage-Grouse and other species, reporting sound levels near the detectors’ noise floor without acknowledging its influence is misleading, although common. Omission of this issue has led to confusion among federal and state agencies, wildlife biologists, industry representatives, and politicians regarding sound levels and sage-grouse management. Further, it risks perpetuating inaccurate data that could lead to inappropriate management. It is essential that researchers report the limitations of their data, and it is important that Greater Sage-Grouse managers understand such limitations.

CONCLUSION

There is strong evidence that Greater Sage-Grouse are sensitive to many forms of human disturbance (Naugle et al. 2011, Wisdom et al. 2011), yet the causal mechanisms linking disturbance to the species’ declines are poorly understood (Crawford et al. 2004). Furthermore, relatively little work has been done to uncover potential threshold levels of disturbance above which it has negative effects (but see Harju et al. 2010, Knick et al. 2013). In analyzing sound levels and count trends at leks within and around an active gas field, we found evidence both for the direct effect of anthropogenic sounds as a causal mechanism for declines in sage-grouse counts and for thresholds of sound ($L_{Aeq} = 31$ dB; $L_{A50} = 26$ dB) above which the number of grouse displaying at leks decreases. We do not know if the declines associated with high sound levels are due to increased mortality or reduced productivity, or if birds simply leave leks with high sound levels and move to other leks. The threshold of 10 dB over background, which has been used to set noise limits in Wyoming and
other states, was based on studies of songbirds (WGFD 2003, Dooling and Popper 2007). Our findings suggest that this criterion is also appropriate for Greater Sage-Grouse in Wyoming, as long as the background level is quantified accurately. Whether this criterion is appropriate for all species remains to be determined. Our findings also suggest that sensitivity to absolute values of sound levels may differ by species. Alternatively, management prescribing a sound level not to exceed $L_{\text{Aeq}} = 31 \text{ dB}$ and $L_{\text{A50}} = 26 \text{ dB}$ could be considered. This approach would eliminate the need for establishing background sound levels, which is often difficult or impossible, but additional study is needed to determine if such thresholds are appropriate in other parts of the sage-grouse’s range and for other anthropogenic sound sources.

Our results highlight the challenges to effective management appropriate for the Greater Sage-Grouse and other sensitive species as human activities such as energy development (gas, oil, wind, solar, etc.), recreation, agriculture, and urban expansion encroach further into wild areas. Species vary widely in their responses to anthropogenic sounds, and the Greater Sage-Grouse appears to be on the sensitive end of the spectrum. Our results highlight the need to study a diversity of species and habitats to improve our ability to predict and mitigate the effects of noise on bird populations.

ACKNOWLEDGMENTS

The Wyoming Game and Fish Department and the Bureau of Land Management’s Pinedale Anticline Project Office funded many aspects of this study. We appreciate the advice and assistance of the WGFD’s Tom Christiansen (retired) and Dean Clause. Jesse Barber, Shan Burson, Matt Maples, Dan Mennitt, Dan Ruthrauff, Mike Schroeder, Carolyn Simms, and Philip Unitt reviewed drafts of this paper and suggested many improvements. We appreciate statistical advice from Anna-Marie Benson, Eric Hintze, and Robert O’Brien. Gail Patricelli and Holly Copeland provided many useful suggestions during our analyses, and Holly Copeland provided information from the University of Wyoming’s Geographic Information Science Center imagery. Alan Rasmussen and Richard Craig assisted with our understanding of the operation and output of the Larson-Davis sound-level meters. We are most thankful to the many individuals, primarily from the WGFD and BLM, who made the many pre-sunrise treks to the leks to count grouse. Without such data, we would not have been able to assess the influence of gas-field noise on the counts’ trends.

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THE STATUS OF LOONS IN IDAHO

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ABSTRACT: This report summarizes and updates the population status and distribution of loons in Idaho. Formerly, the Common Loon bred in northern, central, and southeastern Idaho, probably widely. With only sporadic unsustained nesting in the last half century it has been essentially extirpated as a breeding species, but flocks of up to 200 migrating birds occur in spring and autumn. In spring, concentrations are most frequent in southeastern and south-central Idaho, but in autumn they are widespread. Common Loon migration peaks from mid-April to early May and from mid-October to mid-November. The species winters in numbers of up to 20 on large lakes in northern and southwestern Idaho. The Pacific Loon was not conclusively recorded in Idaho before 1974, but since the mid-1980s a few to dozens have occurred annually in autumn throughout the state, with concentrations of up to 30. Its autumn migration peaks from mid-October to late November. In most years a few now winter, primarily in northern Idaho. There have been a dozen spring and three summer records. Early in the 20th century the Red-throated Loon was recorded rarely in migration but since the 1980s it has been recorded most years, most frequently in autumn. First noted in 1979, the Yellow-billed Loon remains rare. In spring, summer, and winter it is recorded principally from large lakes in northern Idaho, but autumn records are widespread.

In Idaho, loons have received very little attention from ornithologists. The major works on Idaho birds (Larrison et al. 1967:63, Burleigh 1972:1) provide only anecdotal information, with no specific nesting records of the Common Loon (*Gavia immer*) and no records for two other species now known to occur in the state. The Common Loon is considered a “species of greatest conservation need tier 2” and “critically imperiled breeding/imperiled nonbreeding” by the Idaho Department of Fish and Game (IDFG 2017), and a “sensitive species” in the U.S. Forest Service’s intermountain and northern regions (IDFG 2005, 2019). The paucity of information and precarious status calls for a synthesis of data on the breeding status, population dynamics, chronology, and distribution of loons in Idaho. This report summarizes what is known about nesting Common Loons in the state, and the abundance, distribution, and seasonal occurrence of all loon species occurring in Idaho.

STUDY AREA AND METHODS

I attempted to gather and evaluate all information about loons in Idaho. My literature review for Idaho and the adjacent states included the relevant seasonal reports in *American Birds* (AB), *National Audubon Society Field Notes* (NASFN), and *North American Birds* (NAB) from 1961 to 2018, as well as the results of Christmas Bird Counts (CBC) from 1954 to 2018. I compiled data from the Sturts Idaho Bird Records Database (https://ibrc.idahobirds.net/idaho-bird-records-database) and the reports endorsed by the Idaho Bird Record Committee (IBRC; https://ibrc.idahobirds.net/rare-bird-reports), which has reviewed many of the reports of the Red-throated (*G. stellata*) and Yellow-billed (*G. adamsii*) Loons and summer reports of the Pacific Loon (*G. pacifica*). All specific reports of these loon species not in the main text are listed in Appendix 1, available at https://archive.westernfieldornithologists.org/
THE STATUS OF LOONS IN IDAHO

archive/V52/Taylor-loons.pdf. I contacted biologists monitoring or studying loons in Idaho and adjacent states, including those associated with state and federal agencies, the Biodiversity Research Institute, and the Greater Yellowstone Loon Working Group. I used Google Earth Pro to map the Common Loon’s reported nesting sites and seasonal concentrations, the latter based on maximum numbers reported at various sites (Appendix 1). The graphs of seasonal abundance of the Common and Pacific Loons are based on records entered at www.eBird.org through February 2019. For these I went to the “species map” tab for the species, setting the date as March–May or August–November, bringing up all reports for the period, and compiling the number of loons at each site by date. When there were multiple reports on the same date at a “hotspot” and nearby points, I recorded only the highest count at the site for that date. I then tallied all individual counts for each date and plotted them with GraphPad Prism8 (www.graphpad.com).

RESULTS

Breeding/Summer Status of the Common Loon

In the past, Common Loons bred in northern and central Idaho and likely along the state’s southeastern edge (Figure 1, Appendix 1). Owing to the

Figure 1. Locations of recorded breeding of the Common Loon in Idaho. See Appendix 1 for sources.
woeful history of early Idaho ornithology, the historical status of nesting loons in Idaho is opaque. At the end of the 19th century Merrill (1897) found the species common and nesting at Lake Coeur d'Alene, and Tyro (1894) described an adult with young at Priest Lake and courting loons at nearby Blue Lake in northern Idaho. Neither Larrison et al. (1967:63) nor Burleigh (1972:1) listed any specific records of nests, but both authors cited nesting

Figure 2. Locations of concentrations of the Common Loon in Idaho, August–November, by five categories of numbers. Smallest solid dot, 11–20; smallest circle with opening, 21–50; second smallest circle with opening, 51–100; biggest circle with opening, 100–130; square in north, peak of 166; rectangle in south, peak of 212. See Appendix 1 for sources.
THE STATUS OF LOONS IN IDAHO

in parts of northern, central, and potentially southern Idaho (Figure 1, Appendix 1). The Common Loon was observed and collected in summer in southeastern Idaho in the early 20th century (Rust 1917, Taylor et al. 1997). In 1985 and 1986, Fitch and Trost (1985 and C. H. Trost pers. comm.) surveyed 155 lakes throughout Idaho with potential habitat for nesting loons. They observed territorial behavior at Lake Pend Oreille and Cascade Reservoir but did not find any active nests or young. Evers et al. (2010) concluded that the Common Loon had been extirpated from Idaho by the mid-20th century. Any reports on Breeding Bird Surveys from 1966 to 2015 were insufficient for Sauer et al. (2017) to report a trend line for the Common Loon in Idaho. Since the 1980s occasional pairs of loons have nested sporadically for one or a few years in northern, central, and southeastern Idaho (specific records in Appendix 1), but no consistent populations are established, and the species remains effectively extirpated.

Nonbreeding Common Loons, often immatures, occur infrequently in the summer. Fitch and Trost’s (1985) survey of 155 lakes yielded a total of 33 birds, most in northern and southeastern Idaho. Taylor et al. (1999) recorded up to six at American Falls Reservoir, June–July. The ~35 reports via eBird during these two months, of one or two birds, are scattered throughout the state but concentrated at Island Park Reservoir and Henry’s Lake.

Status of the Common Loon Outside the Breeding Season

Autumn. Larrison et al. (1967) considered the Common Loon a regular migrant during October in the north, less so in the south. Burleigh (1972) stated it was a fairly common transient with most records from October, but neither quantified numbers. Hand (1941) considered it common on lakes in northern Idaho, and recorded 9 on Fish Lake 28 October 1928 (Hand 1932). The Common Loon is now known to be a common migrant throughout the state with the largest concentrations on the biggest lakes and reservoirs (Figure 2, Appendix 1). A tally by date of all reports to eBird from August to November shows numbers are very low from August to mid-September, then increase to a peak from mid-October to mid-November (Figure 3).

Winter. Larrison et al. (1967:63) and Burleigh (1972:1) considered the Common Loon to winter in Idaho rarely or in small numbers. The winter reports published in North American Birds, etc., from 1961 to 2017 contain few specific records of the Common Loon. Approximately 20 in winter 1997–1998 was a number higher than normal (NASFN 52:227). December–February reports to eBird are concentrated around Lake Pend Oreille, Coeur d’Alene Lake, and the Snake River near Lewiston in northern Idaho, and C. J. Strike Reservoir and Lake Lowell in southwestern Idaho. Lying at lower elevations, these are large bodies of water that tend to stay ice free much or all of the winter. Lake Pend Oreille had by far the highest concentrations with counts ≥10 in 4 years and a maximum of 20 on 12 December 2008 (Sturts; T. Little and J. Isacoff, eBird). Except for Coeur d’Alene Lake (5 in December 2014; S. Joyce, eBird), no other site had more than 3.

CBCs for all of Idaho from 1955 to 1988 found 1–3 loons in about one-third of the years, with the exception of 13 loons in 1963 (of which 11 were on the Nampa CBC). From 1989 to 2017, loons occurred every year, with a mean of 9.2 per year and a peak of 29 in 2012. This increase coincided with the ini-
tiation of the Spirit Lake, Sandpoint, and Bruneau CBCs, which incorporated sections of Pend Oreille and Coeur d'Alene lakes and C. J. Strike Reservoir. These three CBCs accounted for 174 (65%) of the 266 loons counted during this time. Thus small numbers of Common Loons winter consistently in Idaho, occurring primarily on large lakes and reservoirs at lower elevations and on sections of the Snake River in northern and southwestern Idaho.

Spring. Until late in the 20th century, the Common Loon was reported as regular but occurring in only low numbers. Larrison et al. (1967:63) found it a “regular migrant (only a few individuals) on northern lakes; less regular in south, (regular on Island Park Reservoir, Henry and Gray’s Lake),” occurring in April, while Burleigh (1972:1) listed a few scattered March to mid-May records from across the state. The only large numbers recorded during the 1960s and 1970s were 20 at Lake Lowell and 15 by American Falls Dam in 1976 (AB 30:866). In the 1980s and 1990s large numbers were reported only from Benewah Lake in northern Idaho (up to 20; Sturts) and Twin Lakes Reservoir in southeastern Idaho (peak counts of 150, 100, 80, and 72 from 1987 to 1991; AB 41:463, AB 42:464, AB 44:446, AB 45:474). During this time, the reservoir was closed to fishing until late spring (journals of C. H. Trost). Increasing observations over the last half century now show the Common Loon to be a widespread migrant throughout the state on big and small lakes, rivers, and reservoirs from March to May (Appendix 1). Large groups of loons have been found most commonly in south-central and southeastern

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**Figure 3.** Temporal pattern of Idaho reports of the Common Loon to www.eBird.org, August–November, for all years through 2018. Each dot represents the total number of loons recorded on that date from all hot spots and sites. The trend line is derived by second-order smoothing of four neighboring dots.
Idaho, much less so in central, southwestern, and northern Idaho (Figure 4, Appendix 1), despite birders’ activity in the former areas being much lower. The rate of reports to eBird accelerates rapidly from mid-March to a peak from mid-April to early May (Figure 5).

**Figure 4.** Locations of concentrations of the Common Loon in Idaho, March–May, by five categories of numbers. Smallest solid dot, 11–20; smallest circle with opening, 21–50; second smallest circle with opening, 51–100; biggest circle with opening, 100–110; rectangle, peak of 150. See Appendix 1 for sources.
Pacific Loon

Autumn. The Pacific Loon was first recorded definitively in Idaho on 3 November 1975 when D. Woodby collected a specimen near Viola, Latah Co. (Weber 1978). By 1985 there were but 12 state records, all from between 20 October and 8 December (Taylor and Trost 1987). Autumn sightings increased rapidly starting in 1986 with reports of a few to dozens almost every year and a maximum concentration of 30 (Appendix 1). Autumn reports via eBird, totaling 174 individuals, imply arrival in late September and peak migration from mid-October to mid-November (Figure 6). The Pacific Loon occurs primarily on the same large lakes and reservoirs as the Common Loon, including Pend Oreille, Coeur d’Alene, Cascade, Lowell, C. J. Strike, American Falls, and Island Park, but also occasionally on smaller bodies of still water and the Snake River.

Winter. The Pacific Loon was first noted in Idaho, though without published support, on the Sandpoint CBC in 1954 (Sturts). Thirty-two years would elapse before the next winter record, this from Hayden Lake 2–9 January 1986 (AB 41:305). Subsequently, the Pacific Loon has been recorded most winters. The >30 additional records are mostly from northern Idaho with a few in the southwestern part of the state and one each in south-central and southeastern Idaho (Appendix 1).

Spring. Through the mid-1980s there were no spring records (Taylor and
Figure 6. Temporal pattern of Idaho reports of the Pacific Loon to www.eBird.org, August–November, for all years through 2018. Each dot represents the total number of loons recorded on that date from all hot spots and sites. The trend line is derived by second-order smoothing of four neighboring dots.

Trost 1987), but since 1990, 11 records from 4 March to 21 May have accumulated, of which 7 are from northern Idaho (Appendix 1).

**Summer.** Single individuals have been recorded at lakes Pend Oreille (16 June 2001; C. Swift, eBird) and Coeur d'Alene (4 June 2013; Idaho Survey of the Intermountain Bird Observatory, eBird) in northern Idaho and Daniel's Reservoir in southeastern Idaho (18 June 2020, pers. obs.).

Red-throated Loon

**Autumn:** The first report of a Red-throated Loon in Idaho was based on a specimen from Coeur d'Alene examined by Rust (1913) on 12 October 1912 but apparently not preserved. There are two records from the 1970s (AB 29:89, AB 30:91), and since 1980 the species has been recorded almost annually, beginning in August (Appendix 1). Most records are from northern Idaho, especially Coeur d'Alene Lake, but records are scattered throughout the state.

**Winter:** Red-throated Loons have rarely occurred in this season. Three northern Idaho records are from Hayden Lake (AB 41:305; AB 42:297; AB 44:296) and another is from Coeur d'Alene Lake (NASFN 48:208). In southwestern Idaho two were at Lake Lowell near Nampa through the winter of 1975–76 (AB 30:742) and one was at C. J. Strike Reservoir through the winter of 2010–11 (many obs., eBird).

**Spring:** From 1919 to 1921 birds were recorded around Minidoka 5 sepa-
rate times from 14 April to 20 May (Davis 1935), but only 4 more times in the ensuing century. The subsequent spring records are of single individuals near Moscow in May 1982 (“no details supplied,” AB 36:875), on Hayden Lake, 20 March 1990 (AB 44:446), and on Lake Pend Oreille, 14–16 April 2007 (NAB 61:483) in northern Idaho and near American Falls Reservoir, 2–3 May 1999 (NAB 53:304) in southeastern Idaho.

Summer: In northern Idaho summering Red-throated Loons were recorded at Hayden Lake 17–21 June 1981 and McArthur Lake Wildlife Management Area on 28 June 1982 (IBRC).

Yellow-billed Loon

Since the first report in 1979 (AB 34:182), one or a few have been reported many years. About half the sightings have been submitted to the IBRC, with 9 of 11 accepted. Winter and spring records are primarily from the northern lakes Pend Oreille and Coeur d’Alene, but a few records are from south-central and southeastern Idaho. Autumn records come from southern Idaho and Lake Pend Oreille. The only summer records are of immature birds at Coeur d’Alene Lake on 16 June 2004 (NAB 58:566) and 5 July 2020 (IBRC rare bird report 2020-B-28).

DISCUSSION

The evidence suggests that Common Loons formerly bred on lakes in northern, central, and southeastern Idaho, probably widely, but have been essentially extirpated. Sporadic nesting may have also occurred on reservoirs in south-central and southwestern Idaho. Gap analysis indicates suitable nesting habitat remains in these regions of Idaho (Scott et al. 2002). Nesting loon populations are present in adjacent northwestern Montana (Marks et al. 2016, p. 127), the Greater Yellowstone Ecosystem (V. Spagnuolo pers. comm.), and eastern Washington (Savoy et al. 2019), where a pair nested very close to Idaho in 1925 (Ransom 1929). Hunting, disturbance by human activities, and lead poisoning have been considered possible reasons for this decline (IDFG 2017, V. Spagnuolo pers. comm.). The population trend for the Common Loon on the Breeding Bird Surveys in the Northern Rockies Region is positive, but variation is wide and the credible intervals for the trend estimates straddle zero (Sauer et al. 2017). Efforts to re-establish breeding loon populations have met with some success in Montana, Wyoming, and eastern Washington (Marks et al. 2016, Savoy et al. 2019, V. Spagnuolo pers. comm.). Artificial nesting platforms were placed in Upper Priest, Priest, Pend Oreille, and Coeur d’Alene lakes in northern Idaho as part of the Idaho Bird Inventory and Survey (IBIS) program (IDFG 2005), but without apparent success. There have been no regular surveys there for over 10 years (C. Moulton pers. comm.). Most recent Idaho nests have been found on small isolated lakes (Blue, Thompson Hole, Indian, Bonner) with less human disturbance, suggesting that such disturbance on larger popular lakes is a factor in precluding re-establishment.

The much greater concentration of Common Loons in southeastern and south-central Idaho indicates that these areas are on the main route of the mid-continental breeding populations that cross the Rocky Mountains during spring migration (Evers et. al 2010). In contrast to spring, the larger autumn
concentrations in northern Idaho suggest that the region’s large natural lakes are staging sites for loons moving from nesting areas farther north. Many of the reservoirs in southern Idaho, especially small ones, are often severely drawn down by autumn and provide limited habitat for loons at this time. In both seasons of migration, systematic censusing of key bodies of water, especially in northern, south-central, and southeastern Idaho would clarify their value to migrating loons.

Records of all loon species in Idaho have increased as bird watchers and forums for recording observations have increased. The relative increases of the Common, Red-throated, and Yellow-billed Loons are generally consistent with the increase of effort. Rarely recorded in the early 20th century, the Red-throated has been found increasingly but only in modest numbers, and it remains surprisingly rare in spring by comparison to early records. The lack of any Yellow-billed Loon records before 1979 probably reflects that species’ continuing rarity in Idaho and as well as spotty observer coverage in earlier eras. The dramatic increase of the Pacific Loon since 1980 and near absence of any records before is more enigmatic. The Pacific Loon is not difficult to distinguish. Northern Idaho had Merrill (1897) observing for a few years at Coeur d’Alene Lake, and eminent ornithologists R. L. Hand, E. J. Larrison, and T. D. Burleigh spent decades in this region without recording the species. Southern Idaho suffered from a dearth of observers for much of the 20th century, but C. H. Trost has birded very actively in southeastern Idaho since 1968. It seems remarkable that none recorded a single Pacific Loon if there had been no change in its frequency of occurrence in the state. One possible factor causing this change has been a large increase in the number of reservoirs over time, especially in southern Idaho. Loons use these reservoirs heavily, and their construction in Idaho and adjacent states has probably facilitated loons’ migration through this region.

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LITERATURE CITED

THE STATUS OF LOONS IN IDAHO


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MONITORING THROUGH COMMUNITY SCIENCE: ANNA’S HUMMINGBIRD WINTER RANGE EXPANSION INTO IDAHO

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ABSTRACT: Increased urbanization and supplementary feeding are implicated in driving the expansion of the range of the Anna’s Hummingbird (Calypte anna). In many areas this range expansion has been well described, but the recent expansion of the northeastern limit of the nonbreeding distribution, in winter in Idaho, has not yet been summarized. Using data from the Idaho Bird Records Committee database and www.eBird.org from 1976 through 2020, we collated records for Idaho and supplemented them with data from a community-science program of monitoring by homeowners. Our additional effort to solicit records from the community shows that database records and feeder observations alone underestimate the number of individuals present in the state. Through banding and color-marking of 58 individual hummingbirds at private residences, we documented six instances of Anna’s Hummingbirds returning to a site in successive winters, found a roughly even sex ratio, and found a ratio of adults to juveniles of about 3:1. Anna’s Hummingbird may now be a sparse year-round resident in parts of Idaho.

The range of Anna’s Hummingbird (Calypte anna) spans the west coast of North America from Baja California to British Columbia, into inland areas of California, Arizona, New Mexico, Oregon, and Washington (Clark and Russell 2012). This distribution is a result of an 80-year history of range expansion in the West that has been described through broad-scale analysis and modeling of data from the core of the range (e.g., Clark and Russell 2012, Greig et al. 2017, Battey 2019). While climate change can contribute to widespread distributional shifts for some species (e.g., La Sorte and Thompson 2007, Coristine and Kerr 2015, Prince and Zuckerberg 2015), modeling suggests that for the Anna’s Hummingbird, urbanization, landscaping with non-native plants, and supplementary feeding are extending the species’ realized niche into otherwise unsuitable locations, particularly areas with colder and harsher climates (Greig et al. 2017, Battey 2019). Although these studies describe the extent and potential mechanisms involved in the species’ continuing range expansion, they do not capture a recent eastward expansion of the regular winter distribution of the species into Idaho.

Here, we update the status of the Anna’s Hummingbird in Idaho on the basis of records maintained by the Idaho Bird Records Committee (https://ibrc.idahobirds.net), www.eBird.org, and a community-science program of reporting by homeowners. Apart from updating the species’ status in Idaho, our objectives are to characterize the trend of increasing numbers wintering from October through February since 1976 and to provide additional insights through the homeowner-sighting program and a banding and color-marking study. We document the first six confirmations of winter site fidelity in Idaho, present age and sex ratios of the overwintering population,
and demonstrate that traditional reporting methods can underestimate true abundances.

**METHODS**

**Collating Records**

We summarized Anna's Hummingbird records in Idaho since 1976 from two sources, existing databases and solicited observations.

*Database records.* We extracted reports of Anna's Hummingbirds since 1976 from the database of the Idaho Bird Records Committee. Since 1 March 2011, the IBRC has stopped requesting winter records for the western and northern parts of the state; however, it still requests winter records for central and eastern areas, and also requests summer records for any part of Idaho (https://ibrc.idahobirds.net, S. Sturts pers. comm.). We also compiled sightings from www.eBird.org for October–February, 1976–2020. The IBRC defines “winter records” as those falling between 1 December and 29 February. In our study, wintering hummingbirds often appeared in October and disappeared in February, so we also included October and November records. On the basis of our unpublished data of intra-annual movements of color-marked birds, we applied two main criteria to estimate the number of individuals and avoid double-counting. If observations were more than 1 km apart or more than 30 days apart, we inferred they represented different individuals, and to maintain consistency in protocol and avoid detection biases, we excluded all eBird records that resulted from our banding efforts or had been entered by homeowners associated with the solicited records. We searched Christmas Bird Count data for the same years, but found the records were already represented in the other databases we explored, particularly that of the IBRC.

*Solicited records.* From 2015 to 2020 we actively solicited information from the local communities to understand whether Anna's Hummingbird numbers were under-reported in the existing sources. Using newsletters and posters, we worked with local groups in southwestern Idaho, including the Golden Eagle Audubon Society, the Southwestern Idaho Birders Association, and local bird seed or garden stores, to request reports of sightings. We used email-based birding listservs, the neighborhood social medium “Nextdoor,” the Intermountain Bird Observatory’s webpage, and statewide birding groups on other social media to solicit reports from the rest of the state.

**Banding**

Our efforts at banding and color-marking took place largely in Boise, opportunistically at other locations across Idaho (Table 1). After our call for sightings, we responded to homeowners’ reports of hummingbirds visiting regularly from October to February and requested permission to band and color-mark them whenever possible. With homeowners’ consent, we captured hummingbirds in a Hall trap (see NABC 2019) and banded and uniquely color-marked each individual. For each banded individual we recorded its age and sex, as well as morphometric data. Using correction fluid, we applied a small white mark to the forehead of each banded bird to enable easy tracking,
as these marks are highly visible even from a distance. In addition, we applied a unique color combination of nontoxic permanent ink on the breast of each bird so that individuals could be distinguished (until their next molt). We relied on homeowners’ continued monitoring and reports of color-marked birds to determine whether banding in a given area was complete. If new unmarked birds appeared, we returned to band until all individuals at that home had been banded.

For each location, when possible, we recorded homeowner-estimated arrival and departure dates to make a conservative estimate of the minimum number of individuals present.

**RESULTS**

**Database Records**

The first Anna’s Hummingbird documented in Idaho was in Lewiston, Nez Perce County, 20 November–17 December 1976 (Rogers 1977). Over

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**Table 1** Anna’s Hummingbirds Banded in Idaho, 2015–2020

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<th>Date banded</th>
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<tr>
<td>Boise</td>
<td>16 Oct 2017</td>
<td>Juvenile</td>
<td>Male</td>
<td>K81550&lt;sup&gt;b&lt;/sup&gt;</td>
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<tr>
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</tr>
<tr>
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<td>Male</td>
<td>K81552&lt;sup&gt;d&lt;/sup&gt;</td>
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<tr>
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<td>K81553</td>
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<tr>
<td>Boise</td>
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<td>Female</td>
<td>K81554</td>
<td>3</td>
<td>5.3</td>
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(Continued)
the next 27 years, reports of Anna’s Hummingbirds ranged from zero to two individuals annually (mean 0.64; Figure 1). During the winters of 2004–2011, reports in the IBRC database increased noticeably (range 2–7 individuals annually; mean 3.4; Figure 1). Via eBird, birders reported 151 independent winter records from 1976-2020 (mean 3.4, range 0–29 annually; Figure 1). The majority of these records were near two towns at lower elevations, Boise (832 m) and Lewiston (227 m), and mostly from October to February (Figure 2). Although most frequent in late fall/winter (October–February), reports through eBird now encompass every month of the year (March–September 2006–2014, mean 2.3, range 1–7 annually; 2015–2020, mean 13.5, range 10–23 annually).

Combined data from the IBRC database and eBird illustrate a sharp

### Table 1 (continued).

<table>
<thead>
<tr>
<th>Nearest city</th>
<th>Date banded</th>
<th>Age</th>
<th>Sex</th>
<th>Band number</th>
<th>Fat score</th>
<th>Weight (g)</th>
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<td>K47031</td>
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<tr>
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<td>Male</td>
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<td>—</td>
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<tr>
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<td>Female</td>
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<td>4.8</td>
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<tr>
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<td>Male</td>
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<td>Female</td>
<td>M07790</td>
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<td>Male</td>
<td>K31204</td>
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<td>—</td>
</tr>
<tr>
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<td>Female</td>
<td>K31931</td>
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<td>Male</td>
<td>K31203</td>
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<td>11 Oct 2017</td>
<td>Juvenile</td>
<td>Female</td>
<td>K95455</td>
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<td>4.6</td>
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<tr>
<td>New Meadows</td>
<td>10 Nov 2018</td>
<td>Adult</td>
<td>Male</td>
<td>K47034</td>
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<td>5.3</td>
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<tr>
<td>Twin Falls</td>
<td>29 Nov 2015</td>
<td>Adult</td>
<td>Male</td>
<td>J98802</td>
<td>3</td>
<td>5</td>
</tr>
</tbody>
</table>

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*t, trace; 1, less than ⅓ in interfurcular space; 2, ½ to ⅔ fat in interfurcular space; 3, interfurcular space full

Symmetric molt in primaries, primary 9 most recently grown, rectrices all growing in concurrently.

Symmetric molt in primaries, primary 7 most recently grown, tail finished recent molt.

Tail in symmetric molt, rectrix 2 on both sides growing.
increase since 2011 in both the spatial distribution and numbers of Anna’s Hummingbirds wintering in Idaho (Figure 2). This trend is biased toward centers of human population, whereas many sparsely populated counties lack records of the species.

Solicited Records

During the winters from 2015 to 2020, homeowners reported 36–61 individuals annually, mostly in southwestern Idaho (Figures 1 and 2). Homeowners also reported Anna’s Hummingbirds outside of the period of wintering (October–February), such as an adult male in Boise observed singing regularly at the same location for four consecutive springs (2016–2019), and an adult female photographed in the same neighborhood on 12 June 2020.

Banding

Since November 2015, associates of the Intermountain Bird Observatory and banding colleague Francine Rudeen have banded and color-marked 58 individual Anna’s Hummingbirds across Idaho (Table 1). The majority were in the southwestern part of the state, but some were as far north as Coeur d’Alene, Kootenai County, in the panhandle and as far east as Inkom, Bannock County, in the southeast (Table 1).

Documenting individuals. Our color-banding study measurably increased the number of individuals we were able to document at a given location. Through color-marking and resighting, we identified more individuals at most banding locations than had been reported previously from observations at feeders alone. For example, over a 2.5-week span in November 2015, we identified six different individuals at one location in Boise, when the homeowner’s observations alone, based on the distinct ages and sexes present, suggested only three birds (Table 1). In the most extreme example, on a single morning on 21 November 2016, at a different home in Boise, we banded and color-marked seven birds and observed two additional distinct, unmarked birds for a minimum of nine at one location, when the homeowner’s observations alone suggested just two or three birds (Table 1).

Our banding data indicate that the majority of birds captured in Idaho are adults (Figure 3). Of the 58 individuals captured, 22 were adult males (38%), 20 were adult females (34%), 8 were juvenile males (14%), and 7 were juvenile females (12%) (Figure 3; Table 1). Following Wells et al. (1996) and Pyle (2001; see also Howell 2003), we conservatively aged one female as unknown, as its plumage and bill-corrugation characteristics conflicted.

Site fidelity. On 9 November 2018, we confirmed site fidelity of a nonbreeding Anna’s Hummingbird in Idaho for the first time when we recaptured a bird where we had banded it the previous year in November 2017. In 2019, we recaptured another two individuals at the locations of their banding the previous winter. Although not examples of precise site fidelity, we also recaptured two individuals that were within 1 km, and a third individual within 6.5 km, of locations of their original banding (Table 2). Homeowners did not report these birds during the intervening summer.
DISCUSSION

While the expansion of Anna’s Hummingbird has been modeled comprehensively for the species’ core range (Greig et al. 2017, Battey 2019), these studies were based on data sources that do not represent the trend in Idaho. Our collation of records in freely accessible databases (i.e., eBird and IBRC) shows that this level of documentation adequately detected the trend of Anna’s Hummingbird’s occurrence increasing over time (Figure 1). However, our additional effort to solicit records from the community shows that these databases underestimated the number of individuals present in the area. The solicited homeowner reports and banding effort provided a more complete picture of the number of Anna’s Hummingbirds present in Idaho during the nonbreeding season. By overlaying all sources, we updated our understanding of the magnitude of the numbers of hummingbirds wintering

Figure 1. Number of reports of Anna’s Hummingbird across Idaho during the nonbreeding season (October–February) by year from 1976 to 2020. Sources are records solicited by the Intermountain Bird Observatory (IBO; 2015 to 2019), the database of the Idaho Bird Records Committee (IBRC; 1976 to 2020), and www.eBird.org (1976 to 2020). Anna’s Hummingbird illustration by Bryce W. Robinson.
in Idaho (Figure 2), and gained fine-scale data that may clarify the patterns and processes of the range expansion.

Homeowners are an excellent resource for documenting sightings. However, there are limits to estimating the number of individuals through observations at feeders alone. Because the age and sex of many Anna’s Hummingbirds other than adult males are often indistinguishable in the field, our captures allowed us to document age and sex ratios with a sample size larger than previously reported (Figure 3; Zimmerman 1973, Greig et al. 2017, Batey 2019). Future exploration of age and sex ratios on this and other frontiers

<table>
<thead>
<tr>
<th>Band no.</th>
<th>Date initially banded</th>
<th>Date(s) recaptured</th>
<th>Distance (km)</th>
<th>Age/sex at banding</th>
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</thead>
<tbody>
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<td>9 Nov 2018</td>
<td>0.0 (2018)</td>
<td>Juvenile female</td>
</tr>
<tr>
<td></td>
<td></td>
<td>23 Dec 2019</td>
<td>0.15 (2019)</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>18 Dec 2019</td>
<td>0.72 (2019)</td>
<td></td>
</tr>
<tr>
<td>K47032</td>
<td>7 Nov 2018</td>
<td>29 Nov 2019</td>
<td>1.0</td>
<td>Adult male</td>
</tr>
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<td>K47033</td>
<td>7 Nov 2018</td>
<td>29 Nov 2019</td>
<td>0.0</td>
<td>Adult male</td>
</tr>
<tr>
<td>M07787</td>
<td>25 Jan 2019</td>
<td>17 Dec 2019</td>
<td>0.0</td>
<td>Adult female</td>
</tr>
<tr>
<td>K47030</td>
<td>7 Nov 2018</td>
<td>17 Dec 2019</td>
<td>6.42</td>
<td>Adult male</td>
</tr>
</tbody>
</table>

*Distance between sites of original banding and recapture the following year.
of range expansion may be informative for predicting the pace, success, and outcomes of continued colonization into new regions of expansion as they appear (Rutz 2008).

Through banding and color-marking birds, we also gained a better understanding of the overall numbers of Anna's Hummingbirds in Idaho (Figure 1). Given that in other states the species' range expansion began with incidental sightings of few individuals (Clark and Russell 2012), our comparison of homeowners' counts versus numbers captured for banding implies that there may be a broad gap between apparent and actual numbers of individuals in these cases. In addition, banding allowed us to document six cases of site fidelity in successive years. These show there is some level of interannual survival, suggesting that overwintering in this region, which is much colder than the species' primary range, may not be maladaptive. Future work to more thoroughly document the duration of overwinter stays and survival of individuals will further our understanding of the severity of conditions Anna's Hummingbird can endure, as well as the mechanisms that facilitate its continuing range expansion.

The history of the expansion of Anna’s Hummingbird’s range shows a trend of an increase in overwintering before a breeding population is established (Zimmerman 1973, Clark and Russell 2012). Our research focused on documenting the presence of wintering individuals, which, in conjunction with an increase in the number of summer records, suggests that the Anna's Hummingbird is becoming a regular year-round resident species in Idaho.
Although Battey (2019) did not provide data to illustrate the trend that we describe, he did illustrate (figure 3B) changes to the niche space amenable to Anna’s Hummingbird to include the Snake River plain. This, coupled with increasing numbers in the last ten years, indicates that a resident population may already be present in this area of Idaho. Notably, nesting of Anna’s Hummingbird was confirmed in Idaho, in Boise, for the first time in 2020 (Robinson et al. 2021). This record, along with records of putative hybrids detailed by Rudeen and Bassett (2016), and our documentation of winter site fidelity, suggests that Anna’s Hummingbird may already be breeding in the region undetected at low rates, and the species may now be a sparse resident.

With the indications that a resident population of Anna’s Hummingbird may be developing, or even already present at a small scale, one of the largest questions that remains regarding the winter range expansion into Idaho is where in the breeding range those colonists originated. Further study may reveal the provenance of Idaho’s wintering population and provide additional clarity to the patterns and processes of this continuing range expansion.

ACKNOWLEDGMENTS

Thank you to the many homeowners who graciously invited us to band hummingbirds in their yards, and also to the 281 community scientists who contributed their observations since 2015. Thank you to Francine Rudeen and Carl Rudeen for their assistance with banding and data. Thank you to Susan Wethington and Lee Rogers for providing some bands and tools. Thanks to the Intermountain Bird Observatory’s executive director and research director, Greg Kaltenecker and Jay Carlisle, respectively, for supporting this project. Thanks to Stephanie Coates for her assistance with creating initial figures, and to Robert Miller for helpful review.

LITERATURE CITED

ANNA’S HUMMINGBIRD WINTER RANGE EXPANSION INTO IDAHO


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GRANT FUNDING AVAILABLE

Sierra Foothills Audubon grants money for bird-related science and conservation projects in the northern Sierra and the closely adjacent eastern Central Valley. For more information or to request a grant application, please contact Steve Rose (nevcountybba@gmail.com), Scientific Grants Committee chair, Sierra Foothills Audubon Society.
EVIDENCE OF BROOD PARASITISM AND QUANTIFICATION OF RANGEWIDE OVERLAP BETWEEN THE OLIVE WARBLER AND BROWN-HEADED COWBIRD

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ABSTRACT: Brood parasitism is a fascinating natural history phenomenon that provides a window into the coevolution of antagonistic interactions. Many ecological and evolutionary aspects of brood parasitism remain unknown, and new hosts of brood parasites are still being discovered. We document the second instance of brood parasitism of the Olive Warbler (Peucedramus taeniatus) by the Brown-headed Cowbird (Molothrus ater). Apparent lack of habitat and elevation overlap across the ranges of these two species prompted us to examine how frequently they come into contact. We analyzed >3 million Olive Warbler and Brown-headed Cowbird occurrence records from two open-source repositories, eBird and GBIF, to examine both synchronous and asynchronous locality overlap during the breeding season. We found that the two species were documented together simultaneously in only 3.1% of analyzed instances, but that they co-occurred in similar habitat types and/or at similar elevations at 11.8% of localities analyzed across the Olive Warbler's range. Additional research on aspects of ecology and evolution, such as host selection, the cowbird's diurnal patterns of movement, and the dynamics of intracellular pathogens infecting brood parasites and their hosts, may shed light more broadly on the ecological interactions and mechanisms underlying brood parasitism.

Obligate avian brood parasites lay their eggs in the nests of heterospecific hosts and rely on hosts to raise young. All five cowbird species in the genus Molothrus (family: Icteridae) parasitize avian hosts, and their selection of hosts varies from specialized to generalized (Lanyon 1992, Lowther 2018). The most widespread and generalist cowbird is the Brown-headed Cowbird (Molothrus ater), which lays its eggs in the nests of >220 host species (Brown 1994). It has been suggested that Brown-headed Cowbirds may prefer smaller-bodied hosts, that they may avoid large or aggressive hosts with strong defensive capabilities (Strausberger and Ashley 1997), and that they may select for host nests with a certain number of eggs and/or eggs of a certain size (White et al. 2007, 2009). Cowbirds’ host selection remains challenging to study, and field observations of brood parasitism are a valuable way through which this type of ecological interaction is documented.

On the morning of 2 August 2020, we encountered an adult female Olive Warbler (Peucedramus taeniatus) feeding a Brown-headed Cowbird fledgling near Burro Peak, southwestern New Mexico (32.5930° N, 108.4302° W, ~2370 m elevation; photo on this issue’s inside back cover) in ponderosa pine (Pinus ponderosa) and oak (Quercus sp.) forest. The cowbird fledgling was first seen perched on an exposed oak branch and was located by its persistent, high-pitched begging calls. To our surprise, the cowbird fledgling was fed twice by the adult Olive Warbler over the course of ~3 minutes. Each feeding lasted...
a few seconds before the Olive Warbler resumed foraging. After the second feeding, the cowbird moved from its perch and was seen again approximately 30 minutes later in the same general area, where it was easily relocated by its begging calls. The female Olive Warbler was not seen during the second brief sighting of the cowbird fledgling.

The literature suggests that parasitism of the Olive Warbler by the Brown-headed Cowbird is rare (Lowther and Nocedal 2012, Lowther 2018), and only one instance has been previously documented (Corman and Wise-Gervais 2005). Additionally, this combination seems unlikely given apparent differences in breeding habitat and elevation. This evidence prompted us to ask: How often do these two species co-occur synchronously and/or asynchronously in habitat and/or elevation? We sought to assess the potential for the Brown-headed Cowbird to parasitize the Olive Warbler by assessing their range-wide overlap during the breeding season and by considering aspects of life history that might inhibit or facilitate their interaction.

METHODS

We used specimen records from the Global Biodiversity Information Facility (GBIF; www.gbif.org) and observation data from eBird (www.eBird.org) to quantify breeding season overlap. To obtain GBIF records, we used the package “rgbif” (Chamberlain et al. 2019) to download all Olive Warbler records (doi: https://www.gbif.org/occurrence/download/0033584-200613084148143). To limit our analysis more closely to the Olive Warbler’s range, we limited our selection of Brown-headed Cowbird records to latitudes south of 37° N and longitudes between 85.5° and 114° W (doi: https://doi.org/10.15468/dl.hzxwyz). We excluded observations missing coordinates, with coordinate uncertainty >3 km, observations with unknown “basis of record,” and duplicate records. We obtained all eBird data for the Olive Warbler and Brown-headed Cowbird (downloaded on 21 October 2020), and subset the latter to the same latitude and longitude limits as the GBIF data. We eliminated duplicate records, and, to limit observations resulting from extreme sampling effort, we excluded records covering >3.2 km in distance, areas of >1.0 hectares, and durations >5 hours, generally following best practices for using eBird data recommended by Strimas-Mackey et al. (2020). We then selected data corresponding to the May–July breeding period, merged GBIF and eBird data, standardized latitude and longitude coordinates to a resolution of ~100 m (3 decimal places), and calculated localities where the two species overlap. We mapped overlapping localities onto elevation rasters downloaded with the package “raster” (Hijmans et al. 2020), cropped to study areas of Mexico and the southwestern USA.

All analyses were conducted in R, version 4.0.2, with the RStudio interface (R Core Team 2019). Analysis code is available on GitHub: https://github.com/jlwilliamson/OLWA-BHCO. We emphasize that instances of locality overlap may be affected by quality of input data, filtering criteria, and data-processing choices, and that there are likely many more localities where these species co-occur.
RESULTS

We analyzed a total of 1241 filtered of 44,134 raw Olive Warbler records and 38,638 filtered of 3,005,800 raw Brown-headed Cowbird records from GBIF and eBird. Olive Warblers and Brown-headed Cowbirds co-occurred synchronously in 3.1% of instances, or on 26 of 827 eBird checklists reporting the Olive Warbler. Additionally, we found 1959 total instances of asynchronous overlap at 81 localities during both species’ breeding seasons from at least 1969 to 2020 (Figure 1), representing 11.8% asynchronous locality overlap (81 of 687 distinct Olive Warbler localities). Together, this suggests that Olive Warblers and Brown-headed Cowbirds are rarely reported together but may co-occur infrequently across parts of the Olive Warbler’s range. Localities of synchronous and asynchronous overlap extend from the northernmost to nearly the southernmost limits of the Olive Warbler’s range.

Figure 1. Localities of overlap during the breeding season, ~1969–2020, of the Olive Warbler and Brown-headed Cowbird, calculated with data from GBIF and eBird. Lighter shading represents higher elevations. Pink points indicate localities of simultaneous overlap (n = 26); orange points indicate localities of non-simultaneous overlap (n = 81). Not all points are visible, as some overlay others, and multiple instances of overlap at the same locality are not shown. Turquoise triangles indicate localities of the two documented observations of Olive Warbler brood parasitism by the Brown-headed Cowbird.
COWBIRD PARASITISM OF THE OLIVE WARBLER

(Figure 1), suggesting that the potential for overlap exists range wide and is not limited to specific regions.

DISCUSSION

Our observation from southwestern New Mexico is the second documented case of parasitism of an Olive Warbler by a Brown-headed Cowbird. The first was reported on 14 July 2000 along the Mogollon Rim west of Show Low, Arizona, on a survey for the breeding bird atlas (Corman and Wise-Gervais 2005). Both brood-parasitism observations suggest that the Olive Warbler may be a true host, or species that rears cowbird young, rather than simply a victim, or species that receives a brood parasite's eggs in the nest (Lowther 2018). Alternatively, because fledgling Brown-headed Cowbirds may beg to individuals that are not their foster parents (Ficken 1967, Sealy and Lorenzana 1997), it is possible that the Olive Warbler we observed was not a Brown-headed Cowbird host at all and that it was simply feeding a begging fledgling. Although this explanation is not the most parsimonious, the scarcity of parasitism records in light of overlap at >10% of breeding season localities does raise questions about why the Olive Warbler is infrequently documented as a Brown-headed Cowbird host.

Lowther and Nocedal (2012) hypothesized that Olive Warbler parasitism by the Brown-headed Cowbird is unlikely because of habitat separation despite range overlap. The Olive Warbler favors mid- to high-elevation pine-oak forest (Lowther and Nocedal 2012), while the Brown-headed Cowbird is generally found in lower-elevation riparian areas and disturbed habitats (Lowther 1993, Barnagaud et al. 2015). In some parts of its range, such as the Sierra Nevada of California, the Brown-headed Cowbird has expanded to mixed pine and ponderosa forest with introduction of pack animals and expansion of human settlement (Rothstein et al. 1980, Purcell and Verner 1999, Borgmann and Morrison 2010). In northern New Mexico, the Brown-headed Cowbird occurs across a range of elevations and habitats, including disturbed shortgrass prairie (Goguen and Mathews 2001, Goguen et al. 2005), pinyon–juniper woodland (Goguen and Mathews 2001, Goguen et al. 2009), and mixed conifer forest (Curson et al. 2000). Our findings support that the Olive Warbler and Brown-headed Cowbird do co-occur, or have the potential to co-occur, in habitat and/or elevation more frequently than previously documented (Figure 1).

Despite evidence of habitat and elevation overlap, previous censuses in Olive Warbler habitat in Arizona and New Mexico have not documented cowbirds (Balda 1969, Nocedal 1984). It is possible that previous census efforts simply failed to account for the presence of Brown-headed Cowbirds in Olive Warbler habitat, or that the Brown-headed Cowbird may overlap in habitat and elevation with the Olive Warbler in such low numbers that it is rarely detected. Lack of regular census or birding effort in many regions and more remote areas of the Olive Warbler’s range may also limit simultaneous detections of these two species.

An alternative hypothesis is that the Brown-headed Cowbird is rarely detected in mixed conifer forest because of its unique daily pattern of spatial and temporal habitat use. During the breeding season, female cowbirds may
seek nests asocially in mixed conifer forest at higher elevations and spend only mornings in these host-rich habitats, during which time they are largely silent. Each afternoon, they may commute to disturbed areas at lower elevations (i.e., near human settlements, pastures, and horse stables) where they spend the remainder of the day foraging in large social groups (Rothstein et al. 1984, Goguen and Mathews 2001). In New Mexico, individual female cowbirds have been recorded to commute >18 km one way between breeding and feeding sites (Curson et al. 2000). In the evening, cowbirds may roost communally in groups as large as 1000 individuals, together with other blackbird and grackle species, and roosts may be distant from both breeding and foraging areas (Curson et al. 2000). Although Brown-headed Cowbird abundance and probability of parasitism decrease with increased distance from livestock grazing (Goguen and Mathews 2000), cowbirds’ ability to travel long distances each day between breeding, foraging, and roosting sites suggests high behavioral flexibility for responding to widely dispersed resources necessary for survival and reproduction. It also suggests that cowbirds may be rarely detected in Olive Warbler habitat during certain periods of the day.

Brood parasites tend to lay their eggs in the nests of relatively smaller hosts because larger brood-parasitic young have a competitive advantage over the host’s smaller nestlings (Strausberger and Ashley 1997). It has also been shown that Brown-headed Cowbirds parasitize small (<100 g) species significantly more often than large (>100 g) species (Strausberger and Ashley 1997). Additionally, other small passerines, such as parulid warblers, gnatcatchers, and passerellid sparrows, are common Brown-headed Cowbird hosts (Brown 1994, Lichtenstein and Sealy 1998, Lowther 2018). In this regard, the Olive Warbler (mean body mass 11.0 g) appears to be a suitable host for the Brown-headed Cowbird (mean body mass of females 38.1 g, of males 48.7 g; Dunning 2007). Yet, despite the cowbird’s apparent preference for smaller hosts, its nestlings may grow and survive better in the nests of medium-sized hosts, which provide young with more resources than can smaller hosts (Lorenzana and Sealy 2001, Kilner 2003, Kilner et al. 2004). Cowbirds have been observed to parasitize the nests of species across a range of body sizes that do not result in successful fledging (Hatch 1967, 1971), suggesting that nest parasitism may not always be an adaptive behavior. Taken together, the evidence suggests that host choice involves many factors beyond size.

Novel interactions between brood hosts and parasites, such as the Olive Warbler and Brown-headed Cowbird, have the potential to inform our understanding of a different host–parasite relationship: the coevolutionary “arms race” between hosts and pathogens (Valen 1973). Brood parasites may avoid the burden of host-specific ectoparasites and endoparasites, such as malaria-causing haemosporidian blood parasites, by laying their eggs in the nests of species that have different specialized endoparasite fauna (e.g., Soler et al. 1999). Thus host-specific pathogens may be more abundant in hosts than in brood parasites reared in the same environment. Communities of some endoparasites, such as haemosporidians, turn over at finer spatial scales than their hosts, suggesting that bird hosts may encounter diverse pathogens across their ranges, resulting in spatially varying selection on hosts’ immune systems (Williamson et al. 2019, McNew et al. 2021). Could the Brown-headed Cowbird’s parasitism of novel or rare hosts such as the
Olive Warbler, and possible increased use of habitats where it has not been documented in high numbers previously, such as ponderosa pine forest, represent a mechanism for pathogen-pressure avoidance? It is not possible to rule out upslope range expansion or increased use of forested habitat by the Brown-headed Cowbird as a result of changing temperature regimes, landscape modification, and human influence, which we did not test. However, it is plausible that environmental and anthropogenic factors could be coupled with mechanisms of parasite avoidance, which in turn may be linked to environmental and anthropogenic change. Studying intracellular parasite communities of brood parasites and their interspecific brood parasite hosts represent one lens through which we can approach the evolutionary dynamics of ecological interactions, species distributions, and how both will be affected by future climate change.

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LITERATURE CITED


COWBIRD PARASITISM OF THE OLIVE WARBLER


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A female Olive Warbler (Peucedramus taeniatus) feeding a recently fledged Brown-headed Cowbird (Molothrus ater) in the Burro Mountains of southwestern New Mexico, representing only the second known instance of this combination of brood parasite and host.

Photo by Matthew J. Baumann
NOTES

FIRST DOCUMENTATION OF SUCCESSFUL BREEDING FOR THE ANNA’S HUMMINGBIRD IN IDAHO

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Currently, the Anna’s Hummingbird (Calypte anna) breeds along the west coast of North America from Baja California to British Columbia and east into inland areas of California, Arizona, New Mexico, Oregon, and Washington (Clark and Russell 2012). The current distribution is a result of 80 years of range expansion out of Baja and coastal California, largely facilitated by urbanization, landscaping with non-native plants, and supplementary feeding (Greig et al. 2017, Battey 2019). Pollock et al. (2021), summarizing the recent increase in the numbers wintering in Idaho, illustrated that the species’ range continues to expand. This increase in winter numbers over time mirrors the trend in Arizona in the 1960s, when the establishment of a breeding population followed wintering (Zimmerman 1973, Clark and Russell 2012). Pollock et al. (2021) further suggest that Anna’s Hummingbird may now be a sparse year-round resident in parts of Idaho, primarily the greater Boise metropolitan area. Given the recent trend in Idaho that includes an increase in summer numbers in the past three years (ibid.), the establishment of a breeding population in the state seems possible, even probable.

Rudeen and Bassett (2016) presented evidence implying that Anna’s Hummingbird has already bred in Idaho: the captures of multiple individuals in juvenile plumage in August, a gravid female at the same location in two consecutive seasons, and a putative hybrid male Anna’s × Calliope Hummingbird. Given this evidence, they reasonably expected documentation of nesting to be imminent.

On 14 July 2020, Engle discovered an adult female Anna’s Hummingbird attending a nest containing two nestlings in the front yard of her home in the Warm Springs Mesa neighborhood in southeast Boise (Figure 1). The nest was situated on a horizontal limb of a Rose of Sharon tree (Hibiscus syriacus), 3 m high and only 2 m from the front window of the home. Engle’s home sits at 930 m elevation, in an urban neighborhood adjacent to the foothills, and with a landscaped yard including a fountain, pond, multiple flowering plants, and hummingbird feeders. We monitored the nest daily until the nestlings began to climb from the nest on 17 July (Figure 2). Both young fledged successfully on the morning of 18 July.

Following the fledging of the young, we attempted to capture the adult and fledglings at Engle’s feeders. Unfortunately, we were able to capture only the adult female, which we discovered was banded (see M07787 in Table 1 in Pollock et al. 2021). This female was first captured and banded on 25 January 2019 at a home 0.5 km from the location where she nested at Engle’s home. She was recaptured the following winter on 17 December 2019 at the same location, and is among a growing number of individuals whose site fidelity in Idaho over successive winters has been confirmed (see Table 1 in Pollock et al. 2021). Interestingly, the homeowners did not observe her at this location in either summer 2019 or 2020, despite the proximity to the site of her nesting.

The identity of the male parent of these fledglings is unknown. As is general in
Figure 1. Adult female Anna’s Hummingbird (*Calypte anna*) attending two nestlings in southeast Boise, Idaho, 14 July 2020. This photograph represents the first documentation of nesting of the Anna’s Hummingbird in Idaho.

*Photo by Janice Engle*

Figure 2. The two nestling Anna’s Hummingbirds on 17 July 2020, one day prior to their successful fledging on 18 July.

*Photo by Bryce W. Robinson*
hummingbirds, males of *C. anna* are not involved in brood care (Clark and Russell 2012), so it is not surprising we never observed a male in the area following the discovery of the nest. However, Engle did observe an adult male Anna's Hummingbird at her feeders on 7 July, only 7 days before she discovered the nest. Alternatively, it is possible that the female Anna's was paired with a male Black-chinned Hummingbird (*Archilochus alexandri*), the only hummingbird species that breeds commonly in the Boise metropolitan area. Hybridization between *A. alexandri* and *C. anna* has never been directly documented, although multiple birds that fit the expected appearance of this hybrid have been photographed (see Black-chinned × Anna's Hummingbird at www.macaulaylibrary.org), and there is one specimen record (Banks and Johnson 1961). Nevertheless, from our observations and photographs of the juveniles (Figure 2), we assert that these are not a result of hybridization between *C. anna* and *A. alexandri*. We reviewed photographs of fledglings and juveniles of both species available in the Macaulay Library, the detailed plumage descriptions in Howell (2003), and the photographs of nestlings in figure 5 of Greeney et al. (2008) and compared these to photos of the fledglings at Boise. The Boise fledglings differed from juvenile Black-chinned Hummingbirds in being overall duskier in appearance, particularly in the flanks, which had greenish feathers typical of Anna's. The Boise birds also differed from *A. alexandri* in having densely marked throats and relatively dark heads. Although wing shape was difficult to judge in our photos, we noted that the inner primaries were not appreciably narrower than the outer primaries, as is seen in the Black-chinned, and appeared to fit the shapes expected of *C. anna*. Additionally, the shape of the exposed primaries differed from those shown in figure 5 of Greeney et al. (2008), appearing relatively straight in the Boise birds and not substantially curved as in the Black-chinned nestlings. Overall, the plumage of the Boise fledglings seemed to fit the Anna's Hummingbird, and did not appear to be intermediate between the two species, as would be expected in first-generation hybrids.

The timing of this nesting is notable. Throughout its native range, the Anna's Hummingbird breeds from December to May (Clark and Russell 2012). At the margins of its distribution in coastal Washington and British Columbia, nesting begins slightly later, from January to February (ibid.). By taking the average periods of incubation and nestling (16 and 20 days respectively; Clark and Russell 2012) and backdating from the date of fledging on 18 July, we estimate the nest to have been initiated on 12 June, an atypically late date by comparison to the dates reported elsewhere throughout the species' range. Anna's Hummingbirds often raise multiple broods in a single breeding season (Scarfe and Finlay 2001), and this nesting could represent a second brood. The phenology we observed in Idaho is matched by the first reported breeding of the species in southwestern Canada in 1958, which also occurred in July (Campbell et al. 1990). More reporting and continued focus on the species' phenology of nesting at the margins of its range may uncover if the schedule in these two instances is typical for extralimital breeders.

This first documented successful breeding of the Anna's Hummingbird in Idaho provides a marker on the chronology of the species' range expansion into the state. This chronology has thus far been well documented through the methods outlined by Pollock et al. (2021). Continued focus and documentation of the increase in numbers, as well as of further instances of breeding, will provide fine-scale details to the ongoing range expansion experienced by this species since the urbanization of the West accelerated in the middle of the 20th century. Investigations into the provenance of Anna's Hummingbirds wintering around Boise will provide further clarity to the pattern of expansion into Idaho, and elsewhere.

We thank David Towner who initially reported M07787, the nesting female, in his yard in December 2018, through the Intermountain Bird Observatory's community-science program of hummingbird reporting. We also thank Jay Carlisle, Daniel D. Gibson, and Chris Clark for their helpful reviews of the manuscript.
LITERATURE CITED


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APPARENT COMMENSALISM OF A RED-TAILED HAWK AND BADGER

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Hunting associations between the American Badger (Taxidea taxus) and Coyote (Canis latrans) have been well documented (Aughey 1884, Hawkins 1907, Cahalane 1950, Lehner 1981, Kiliaan et al. 1991, Minta et al. 1992). Coyotes have been observed capturing rodents flushed out of burrows by badgers (Minta et al. 1992).

Accounts of badgers being exploited in a similar fashion by birds of prey are not well documented. Phillips et al. (1964:23) wrote, without specific reference, that in Arizona it is “well known” that both Ferruginous (Buteo regalis) and Red-tailed (B. jamaicensis) hawks associate with badgers to capture flushed prey. In West Texas, Wauer and Egbert (1977) described a Harris’s Hawk (Parabuteo unicinctus) that followed a badger overturning cow dung piles. Though feeding was not actually observed, they believed the badger and hawk were seeking insect prey found beneath the piles. In this note I describe another possible instance of hawk–badger commensalism.

At 13:20 on 8 March 2020, while driving on Gartersnake Road, 0.4 km south of Carlsbad Caverns National Park, New Mexico, I observed a Red-tailed Hawk at approximately 200 m away flying toward me. I am accustomed to seeing this species flying overhead, often gliding in large circles, searching for prey, so the hawk’s movement on this afternoon was unusual in that it was flying slowly, not more than 5 m above the ground and in a straight line perpendicular to the road on which I was driving.

I stopped my vehicle and watched the bird approach the roadway. When it was approximately 20 m from the road, an adult badger appeared on the road shoulder about 25 m beyond my vehicle. The badger was traveling in front of the hawk and in the same direction. The badger moved to the middle of the paved roadway and stopped. It turned its head and stared at me, perhaps because it heard the running engine. (Before the badger appeared, I had stopped to view the hawk from the vehicle.) At the moment the badger halted, the hawk arrived at the road shoulder and immediately perched upon a fence post within 7 m of the badger. The badger glanced backward at the hawk, then looked back at my vehicle. After about 10 seconds, the badger trotted off the road and continued heading southwest in a zigzag pattern.

Once the badger left the road, the hawk resumed flying 5 m above the ground, staying approximately 20 m behind the badger. The hawk continued following the badger for at least another 150 m, until I could no longer see the badger with binoculars. The hawk continued flying slowly in the same direction until it disappeared below the horizon.

Annual monitoring of a research site brings me to Carlsbad Caverns National Park for three consecutive days each year. Although I have observed red-tails in the area on every visit, this was the first time I have observed a live badger. The site lies at the edge of the Chihuahuan Desert. Creosote bush (Larrea tridentata), ocotillo (Fouquieria splendens), lechuguilla (Agave lechuguilla), and Torrey yucca (Yucca torreyi) are conspicuous plants in the immediate area. In my experience, vehicles traverse Gartersnake Road only a handful of times each day. On 8 March a slight breeze persisted for most of the daylight hours. The sky was clear and the maximum temperature on that day was 18°C (https://w2.weather.gov/climate/xmacis.php?wfo=maf.NOAA).

On the basis of the many observed coyote–badger interactions, as well as the observation of Wauer and Egbert (1977), I hypothesize that the Red-tailed Hawk
was following the badger on the chance the badger might flush prey, presumably while excavating a rodent burrow. This seems the most likely explanation for the hawk’s behavior. At no time did the badger seem disturbed by the hawk’s proximity, nor did the hawk behave in an aggressive manner toward the badger. An adult badger can weigh up to 12 kg and is known as an aggressive predator (Reid 2006). This may explain why I was unable to locate any records of Red-tailed Hawk predation on badgers.

This account may stimulate other observers to focus on the kind of unusual red-tail flight behavior I have described. An actual observation of a badger flushing prey that is then captured by a Red-tailed Hawk would be an interesting example of vertebrate commensalism. Beyond the specific instance I describe, any kind of unusual animal behavior may reflect patterns that seem novel but, in fact, may be significant to the survival of one or both species.

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BOOK REVIEW


A biological atlas maps the distribution, and sometimes the abundance, of a group of species (most commonly breeding birds) within a defined geographic area during a set time period. These atlases have dramatically advanced distributional studies over earlier ones based on randomly collected specimens or anecdotal observations by attempting to obtain comparable coverage within each (or a random sample) of the equal-sized blocks in an atlas grid overlain on the study area. The first breeding bird atlases began in Britain and Ireland in the mid-1960s (Lord and Munns 1970, Sharrock 1976). The concept quickly spread throughout the world but has been applied mostly in the northern hemisphere (Dunn and Weston 2008). The first atlas project in North America, for two counties in Maryland, was initiated in 1971 (Klimkiewicz and Solem 1978), but most subsequent work has been conducted at larger scales. From 1979 to 2018, atlases have been completed for 9 of Canada's 13 provinces and territories, of which 6 have completed second atlases. They have been completed for 41 of the 50 United States, of which 18 have completed second atlases (Beck et al. 2018).

By contrast, California has taken a county-by-county approach. As of 2020, atlases have been initiated or completed for all or part of 22 of the state's 58 counties; most of which are coastal, and concentrated around San Francisco Bay. Of the 22, 16 have been published in one form or another (Roberson and Tenney 1993, Shuford 1993, Burridge 1995, Shuford and Metropulos 1996, Gallagher 1997, SAS 2001, Berner et al. 2003, SFFO 2003, Unitt 2004, Hunter et al. 2005, Bousman 2007, Glover 2009, Richmond et al. 2011, Rippey 2014, Allen et al. 2016, volume reviewed here). Field work for a second atlas has been completed in Sonoma County, initiated in Sacramento and Santa Cruz counties, and is in the planning stages in Marin County. In three counties field work for first atlases was completed over 25 years ago, but the results have yet to be published. Two of these, however, plan to publish results of their first and second atlases in the same volume. Three atlases were initiated (two long ago, one fairly recently) but never completed with no current plans to do so.

California's latest published atlas, for Nevada County, traversing the northern Sierra Nevada, is organized by five chapters. Description of Nevada County (9 pp.) has sections on location and population, geology, major geographic features, climate and rainfall, major roads, land use, major habitat types, and elevation zones. At 958 square miles, this sparsely populated county (97,182 persons in 2012) is relatively small in size, stretching about 68 miles west to east and from 9 to 26 miles north to south. Elevations range from 250 feet in the Sierra foothills on the west slope to three peaks slightly over 9000 feet at the Sierra crest to 5200 feet on the east slope at the county's border with the state of Nevada. Land in the county is 63% private, 35% public (mostly in central and eastern portions and at higher elevations).

Supporting this information, to a limited degree, is a color-shaded relief map on the inside front cover showing the grid of numbered atlas blocks for Nevada County in relation to surrounding counties and the western edge of the state of Nevada. The only other landmarks shown are major highways and the location of the county's three incorporated cities (Nevada City, Grass Valley, Truckee). Despite inclusion of a gazetteer, those not familiar with the county will have difficulty locating place names mentioned in the text given other towns, major rivers, lakes or reservoirs, prominent mountain peaks, state parks and wildlife areas, or other geographic features are not
BOOK REVIEW

shown. A map showing these would have been extremely helpful, as well as maps depicting elevation contours, the distribution of major vegetation types, and the six elevation zones discussed.

The county’s climate varies from Mediterranean on the west slope, with warm to hot, dry summers and wet, cold, rainy winters, to continental on the east slope, with cool to warm and dry summers with occasional thunderstorms and very cold, snowy winters. Precipitation, mostly as snow, is highest on the upper west slope. “Rainfall” (presumably “precipitation” including snowfall) data are provided for the five years of field work on which the atlas is based, 2014–2018, for two locations, one each on the west and east slopes. These data show drought during the first two years and precipitation over 200% of “normal” in the winter preceding the fourth year. It would have been valuable to also present precipitation data for the period directly prior to the atlas period, as the effects of drought are cumulative and California experienced a severe drought starting in the winter of 2011–12, i.e., two years prior to initiation of the field work.

The authors describe the 12 major habitats/plant communities thoroughly, but scientific names of the trees and shrubs mentioned are lacking. Defined by elevational ranges and dominant plant communities, six elevation zones are also described, and representative birds or species reaching their upper or lower elevational limits within each zone are listed. For birds, scientific names are lacking at their first mention but included later in the species accounts.

The chapter Methods includes sections on the schedule of field work, maps and grid, blocks, abundance estimates, data forms, data management, and adequacy of coverage. Methods generally followed those of many other county atlas projects in California and are clearly described. The county was overlain with a UTM grid of 126 5-km by 5-km atlas blocks; of these, 16 edge or partial blocks were either so small that they were not surveyed or were subsumed into adjacent full blocks. Of 111 blocks judged worthy of coverage, 110 were surveyed from 2014 to 2018. Atlas participants were provided digital maps with a variety of options for backgrounds, including Google Hybrid or Terrain, open street maps, and topographical maps, which could be panned, zoomed in and out, and printed for any block or part of a block. Criteria defining categories of evidence of breeding followed the recommendations of the North America Ornithological Atlas Committee (NORAC; Laughlin et al. 1990).

Observers initially recorded breeding data on paper field cards, or on comparable Excel spreadsheets from data recorded in field notebooks, which were submitted at the end of the breeding season. Drawbacks of adding data to field cards as the season progressed were the loss of some information on breeding phenology when lower-level codes were replaced later by higher ones. But in some of these cases, the date of the original, rather than the later, observation was retained. After several field seasons, observers were encouraged to use www.eBird.org to record atlas data, and many did so. Submitting atlas data by eBird avoided the problems discussed for field cards but required that each checklist be entered into the atlas database and that some of eBird’s breeding codes be converted to the standard NORAC codes. Most data were entered in a database created for the project and double-checked for accuracy, with data on block survey time and other metrics of block coverage tracked by spreadsheet.

Because other atlas projects have found that participants are uncomfortable estimating bird abundance, the Roses themselves, as project leaders, “censused” each atlas block on at least two occasions to assess habitats and gain a general impression of the abundance of each species. They then made a subjective, order-of-magnitude estimate of the number of breeding pairs of each species in each atlas block by assigning each species to one of three categories, 1–9, 10–99, or ≥100 pairs (Allen et al. 2016). Next, they estimated the range of the total number of pairs of each species in the entire county by summing the minimum and the maximum estimates for all
blocks occupied by a species. Prior to this, they made further adjustments toward
the lower end of the range estimate for an unspecified number of species that had
numerous blocks with estimates of 10 to 99 pairs and were judged to not contain
many more than 10 pairs per block. The overall subjective approach to estimating
abundance with further undocumented adjustments does not instill much confi-
dence that this methodology will provide reliable estimates of population change if
repeated in the future. This is a problem faced by all atlas projects and one not easily
overcome without point counts or other quantitative methods that allow rigorous
statistical analyses (Beck et al. 2018, McCabe et al. 2018), which may not be practical
for most county-level atlases. Many atlas projects are now relying on eBird checklists
for recording data, but it appears too early to evaluate whether such data will provide
robust estimates of population change when atlases are repeated.

To assess adequacy of coverage, the project initially set an arbitrary goal of survey-
ing each block for 30 hours and confirming breeding for 30 species. After two years
it became apparent that it would not be possible to do so for all blocks, particularly
those at mid-to-high elevations on the west slope that are dominated by granite
and difficult to access. Ultimately, however, the authors assessed coverage of each
block empirically on the basis of its habitat diversity, the total survey hours to date,
the potential for additional species or confirmations, and the difficulty of access
(Shuford 1993). By these criteria, just 9 of 110 blocks were considered inadequately
covered at the end of the project. But in the atlas these nine blocks are not identified,
making it difficult for the reader to assess whether some species’ observed distribu-
tions have been influenced by limits to survey coverage. It also was unclear if any
special surveys for hard-to-detect or nocturnal species were undertaken. Were any
owing routes covered, or were nocturnal species the purview of just those covering
specific blocks? Regardless, the project leaders considered a sixth year of coverage
in 2019 but decided against it after an evaluation of data from prior years suggested
little would have been gained by doing so.

The bare-bones chapter Results includes just one page of text with sections on
the number of breeding species, species gained and lost, and most abundant or
widespread species. In addition, three tables rank the top ten species with respect
to abundance, the number of confirmed blocks, and the number of blocks where
detected; a fourth table lists the number of blocks in which the most common
representatives of 10 widespread families were detected. The atlas recorded 173
species with either possible, probable, or confirmed evidence of breeding. Of these,
155 species were confirmed breeding in the county, 9 for the first time. The authors
were appropriately conservative in not considering the Cinnamon Teal (Spatula
cyanoptera) and Sandhill Crane (Antigone canadensis) confirmed breeding in the
county because such evidence was found in blocks straddling the county line but the
birds’ natal county could not be determined. Similarly, because families of Clark’s
Nutcrackers (Nucifraga columbiana) can move long distances after fledging and the
adults may feed their young for many months, observations of fledglings or feeding
young in seven atlas blocks were judged not sufficient to confirm the nutcracker’s
breeding in the county, though it almost certainly occurred. Two species, the
Yellow-headed Blackbird (Xanthocephalus xanthocephalus) and Willow Flycatcher
(Empidonax traillii), which bred in the county as recently as 2006, were considered
extirpated as breeders during the atlas period.

The chapter Reading the SpeciesAccounts (4 pp.) has sections on nomenclature,
taxonomy, and subspecies; bird seasons; relative abundance; conventions and com-
ments; and content of the species accounts. The species accounts (346 pp.) constitute
over 90% of the main body of the book and are well researched and informative,
representing the book’s most noteworthy contribution. The accounts make the most
of the atlas data and provide context from other relevant studies. Each of the 167
species accounts includes a map showing breeding distribution and a set of standard
sections (supplemented by four tables): summary, seasonal status, breeding distribution and habitat, breeding phenology and natural history, local and regional trends, and abundance (atlas data, other data). The atlas maps are smaller versions of the map on the inside front cover minus the block numbers and location of the three cities; I had to squint to read the numbers of the major highways. The species accounts are followed by two appendices (shorter accounts for six species of unconfirmed but possible breeders and four species of former breeders), a gazetteer, literature cited, and an index of bird names.

The brevity of the Results, and lack of a discussion, represent an opportunity for a further publication synthesizing the data in this atlas. Graphs could depict how many species fell within various categories of abundance or the number of blocks occupied. Important patterns of distribution could be discussed, and relevant literature could be compared. What species, or proportion of species, spanned the entire county west to east, or occurred mainly on the west slope, east slope, both slopes with a gap at higher elevations, or both higher elevations and the east slope? What might explain such patterns of distribution (perhaps occurrence with respect to key habitat types or the six elevation zones described)? How did patterns of distribution in Nevada County compare to what is known for other counties in the Sierra? How representative is the avifauna of Nevada County of the relatively low elevations of the northern Sierra (north of Sonora Pass), and how is it different or similar to that of the higher-elevation southern Sierra? The basis for such discussion can be found in the species accounts and the distribution maps. Did the precipitation patterns during and immediately preceding the atlas period influence the distribution or abundance of wetland-dependent or other species?

The authors are to be commended for tackling an atlas in a county with few observers, documenting the breeding avifauna of an area with a limited historical record, and for publishing a substantial book in the year following the completion of field work. A troubling trend in county atlases in California as a whole is the often long lag from completion of field work to publication, which is being partly offset by a recent trend of providing online access to preliminary maps during atlas field work and/or final maps after field work but before publication. Another unfortunate aspect of the county-by-county approach is the difficulty of assessing changes in breeding distribution across any of California's ecoregions, such as the Sierra Nevada, or the entire state, given the many counties without atlases and the long time over which atlases have been completed. The county atlases have been undertaken largely by local Audubon Society chapters, local or regional bird clubs, and highly motivated individuals. I hope that at some point atlas work in the state will be embraced by larger conservation and research organizations and government agencies so field work for an atlas for all, or large parts, of California can take place at the same time. In the meantime, there is much merit in fine-grained atlases at the county level, which might not be possible if the entire state were surveyed concurrently, given its large size and extensive remote areas. Hence the Nevada County atlas will be of great value for those interested in the breeding avifauna of the Sierra Nevada or those planning a first or second atlas for their county in this digital age.

LITERATURE CITED


BOOK REVIEW


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W. David Shuford
DO NUTCRACKERS USE AUTOMOBILES AS NUTCRACKERS?

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While driving through Yellowstone National Park, Wyoming (at 44.9° N, 110.5° W, 2122 m elevation), on 22 April 2019, I came across a Clark’s Nutcracker (Nucifraga columbiana) standing in the opposing lane of traffic. The nutcracker had a live Western Tiger Salamander (Ambystoma mavortium) in its bill. Intrigued by the scene, I pulled into a nearby roadway turnout and began to photograph this interaction between predator and prey.

A nutcracker preying on a salamander is not in itself unusual. After all, salamanders living in Yellowstone National Park come out from hibernation in April and migrate to ponds for breeding. An opportunistic predator like Clark’s Nutcracker may take advantage of such a situation because salamanders then become accessible. In fact, several avian species that do not normally feed on vertebrates have been reported preying on salamanders elsewhere: the Tufted Titmouse (Baeolophus bicolor; Hendricks and Hendricks 1985), Hermit Thrush (Catharus guttatus; Coker 1931), and American Robin (Turdus migratorius; Wilson and Simon 1985). Clark’s Nutcracker has been reported preying on various vertebrates, moreover, as small as tadpoles (Pilliod 2002) and as large as adult toads and ground squirrels (Mulder et al. 1978). However, what I subsequently observed was unusual:

• Before a motor vehicle came into sight, the Clark’s Nutcracker placed the live salamander near the center of the traffic lane.
• The nutcracker flipped the salamander onto its back (see photo on this issue’s outside back cover).
• The nutcracker then released the salamander and flew to a roadside tree.
• As a motor vehicle approached, in the same lane as the now supine salamander, the nutcracker remained perched.
• After the motor vehicle passed over the salamander without striking it, the nutcracker flew down to the road surface.
• The nutcracker repositioned the salamander near the original spot, again turning it on its back, and then flew back to its observation perch.

The nutcracker repeated this behavior over a period of approximately 5 minutes, during which motor vehicles passed three times. After the third pass, the nutcracker flew down to the road surface, retrieved the still living prey, and flew off. Each time the bird repositioned the salamander on the road surface, the nutcracker spent time standing over it. It then returned to its observation perch before the next approaching vehicle came into sight. Once a vehicle had passed, the nutcracker was quick to fly down and take possession of the salamander once again. I never observed the bird drop the salamander from height. Rather, it repeatedly placed the salamander on the pavement. (It is possible that the nutcracker dropped its prey prior to my observation.)

Such a sequence of behaviors might suggest that the nutcracker was employing the road’s hard surface and the sporadic flow of traffic as a means to “crack open” the salamander, thereby exposing the prey’s edible parts. So the question posed by this observation, which parallels the one posed by Maple (1974) in this journal 47 years ago: “Do nutcrackers use automobiles as nutcrackers?”

Some of the best documented examples of avian resourcefulness and adaptability (i.e., avian intelligence) have focused on members of the family Corvidae. Corvids
consistently demonstrate intellectual skills superior to those of many other birds, and in many cases comparable to those of primates (Emery 2004). One notable example is that of New Caledonian Crows (Corvus moneduloides) that fashion and then use hooks as tools to retrieve food items otherwise inaccessible (Hunt 1996). Equally impressive are reports from Japan of Carrion Crows (Corvus corone) that deliberately place walnuts in the path of soon-to-be moving automobiles so that the tires will crush the nut's hard shell, exposing its edible parts (Nihei 1995). Nihei and Higuchi (2001) suggested, moreover, that the use of moving vehicles as “nutcrackers” by Carrion Crows followed the precursory behavior of dropping walnuts from height onto hard surfaces to crack them open. It was then by chance that some of the dropped walnuts were inadvertently run over and cracked open by moving vehicles. Witnessing such an event might have stimulated a Carrion Crow to go from simple food-dropping to deliberately using vehicles as nutcrackers.

In Davis, California, Maple (1974) reported a single observation of an American Crow (Corvus brachyrhynchos) that dropped a walnut on a well-traveled street. Wondering if the crow expected a vehicle to crush the walnut, Maple (1974) asked: “Do crows use automobiles as nutcrackers?” He suggested that they do, starting a discussion as to whether or not avian species are capable of using automobiles as nutcrackers. In support of Maple’s suggestion, Grobecker and Pietsch (1978) reported observing a single crow in Long Beach, California, drop palm fruit from height onto a busy residential street. Even though the palm fruit broke into two pieces upon impact with the pavement, the crow flew to an observation perch and waited for one of the pieces to be run over by a vehicle. Only then did the crow fly down to the roadway and retrieve the now shattered fruit.

To exemplify corvid intelligence by citing moving vehicles as “nutcrackers” is not without controversy. Cristol et al. (1997) contended that their own observations of crows in Davis, California, did not support Maple’s (1974) and Grobecker and Pietsch’s (1978) conclusions, but rather that crows’ perceived use of vehicles as nutcrackers was simply an incidental byproduct of the crow’s food-dropping behavior: They disagreed that crow behavior has developed beyond the level of food-dropping.

Though I did not witness the nutcracker dropping its prey, it might have done so prior to my observation. But dropping this soft-bodied prey from height would have been ineffective at “cracking” it open, given the tough, pliable skin of a salamander. Dropping prey from height onto a flat surface such as a roadway may be an effective means to stun or stress soft-bodied prey, but employing such behavior to expose the innards would be futile. In addition, even though such behavior has been documented in at least two dozen avian species, including at least nine of Corvus (Cristol and Switzer 1999), I’m not aware of a previous report of Nucifraga engaging in such behavior.

Certainly a nutcracker’s bill can puncture a salamander’s skin. But, lacking a hooked bill adapted for tearing flesh, a nutcracker might not be able to tear the skin sufficiently to expose the edible parts. The weight of a motor vehicle would surely do the job, however messily. And an automobile’s quick dispatch of a salamander would also substantially reduce the amount of toxins a salamander might excrete if a nutcracker continued to toy with it. (I identify the whitish droplets evident on both the salamander and the nutcracker as excreted toxin; see photo on this issue’s outside back cover).

One possible explanation for the behavior observed is that this particular Clark’s Nutcracker had previously witnessed salamanders crossing this section of road and, consequently, getting “cracked open” by passing vehicles. Perhaps this particular Clark’s Nutcracker had even fed on such serendipitously available prey. That experience might have stimulated this bird to purposely place the salamander supine on the road surface in an attempt to convert a chance event into a determined one—a conjecture I recognize is derived from a single and all-too-brief field observation.
At minimum, my report adds to a long-running conversation about birds’ use of artifacts as tools. But if future observations or studies of nutcrackers confirm this capability, then this observation becomes another record of what Maple (1974:98) eloquently described as “an ingenious adaptation in response to the intrusion of man’s technology.” And it does so for a member of the family Corvidae yet to be acknowledged as capable of such an adaptation.

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LITERATURE CITED


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THANK YOU TO OUR SUPPORTERS

The past year has been challenging to all of us on many levels. WFO’s budget took a serious hit as the coronavirus pandemic compelled cancellation of our annual conference and multiple field trips. But our membership compensated with extraordinary generosity, for which we are deeply grateful. The board of Western Field Ornithologists and the editorial team of Western Birds thank the following contributors for their financial support in 2020.

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The population of the Greater Sage-Grouse has declined dramatically because of multiple factors, of which one is large-scale extraction of fossil fuels from under its habitat. Besides the habitat loss, fragmentation, and disturbance resulting from this industry, the noise that drill rigs generate is also a factor. In this issue of Western Birds, Skip Ambrose, Christine Florian, Justin Olnes, John MacDonald, and Therese Hartman evaluate the best methods for quantifying noise pollution in the sagebrush steppe of Wyoming and evaluate its effects on the Greater Sage-Grouse through 20 years of monitoring at leks. They confirmed that noise has an adverse effect on the grouse and identified the threshold above which the effect is significant.

A female Olive Warbler (Peucedramus taeniatus) feeding a recently fledged Brown-headed Cowbird (Molothrus ater) in the Burro Mountains of southwestern New Mexico, 2 August 2020, representing only the second known instance of this combination of brood parasite and host. This combination thus seems surprisingly rare in spite of the Olive Warbler’s apparent suitability as a host. In this issue of Western Birds, Jessie L. Williamson and Matthew J. Baumann explore the use of large public databases, eBird and GBIF, to quantify overlap between the Olive Warbler and Brown-headed Cowbird in time and space.
“Featured Photo” by © P. Ross Gorman of Cody, Wyoming: Clark’s Nutcracker placing a salamander on its back in the middle of an active roadway, 22 April 2019, Yellowstone National Park, Wyoming. Note on both the salamander and the nutcracker the whitish droplets of toxin excreted by the salamander, suggesting that the salamander’s skin had already been pinched or pierced. But the salamander is otherwise intact and actively defending itself.