Article

Annual Mortality Limit for Four Gull Species in the Atlantic Flyway

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Abstract

We estimated the allowable annual take of great black-backed gulls Larus marinus, herring gulls L. argentatus, ring-billed gulls L. delawarensis, and laughing gulls Leucophaeus aterciillia in the U.S. portion of the Atlantic Flyway to help meet human safety and resource management goals. Gulls can pose a serious threat to aviation, negatively impact other colonial-nesting migratory bird species, and conflict with other human activities. We estimated an annual take limit using a model that incorporated intrinsic population growth rate, minimum population size, and a recovery factor for each species. We estimated intrinsic population growth by combining allometric with life table approaches. We used the recovery factor to restrict the take level of the great black-backed gull beyond that of the other species because of poor data quality and concern about its population status. The herring gull was the only species with comprehensive demographic data. Population sizes used in estimating potential take limit varied greatly among the four species, but estimates of intrinsic population growth rate were similar (range 0.118 to 0.197). The annual potential take limits for the four gull species were 7,963 for herring gulls, 2,081 for great black-backed gulls, 15,039 for laughing gulls, and 14,826 for ring-billed gulls. Comparing average annual take from 2012–2019 to our modeled potential take limit, overharvest has not occurred for great black-backed and laughing gulls, occurred once every 8 y for ring-billed gulls, and occurred over half the time for herring gulls.

Keywords: allowable take; Larus sp.; gulls; Atlantic Flyway; intrinsic growth rate; allometric models; life table models

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Introduction

Gulls (family Laridae) are ubiquitous in the eastern United States and are especially abundant in the Great Lakes and along the Atlantic seaboard. Gulls provide valuable ecosystem functions (Morris et al. 1992; Sekercioglu 2006) and are popular with bird watchers because of their abundance and many varied plumage characteristics; however, there is often conflict between gulls and human activities and needs. Gulls can negatively impact sensitive wildlife species via competition for space and by predation (Hatch 1970; Becker 1995; Russell and Monteverchi 1996; Pius and Leberg 1997; Scopel and Diamond 2017). In high densities, gulls negatively alter water quality in parts of the Great Lakes and along the Atlantic Coast (Fogarty et al. 2003; Converse et al. 2012; Winton and River 2017). Most
problematic is that gulls are involved in aircraft strikes at and near airports, resulting in millions of dollars in damage and risk to human injury and loss of life (Burger 1985; DeVault et al. 2018; Dolbeer et al. 2021).

The Migratory Bird Treaty Act and accompanying case law and regulatory framework give the U.S. Fish and Wildlife Service (USFWS) responsibility for issuing depredation permits authorizing take of migratory bird species. When the USFWS permits management of conflict bird species, the permitted actions can be nonlethal (such as hazing or habitat management to deter or exclude) or lethal. Although nonlethal actions are preferred, sometimes lethal take of gulls is necessary for addressing conflicts (Dolbeer et al. 1993; Nugent et al. 2008). It is the responsibility of the USFWS to ensure that take is within sustainable limits for bird populations and that it does not impact long-term conservation of species. Great black-backed gulls Larus marinus, herring gulls L. argentatus, ring-billed gulls L. delawarensis, and laughing gulls Leucophaeus atricilla receive some of the most requests for depredation permits in the Atlantic Flyway. Take of the four gull species occurs year round in the Atlantic Flyway, and from 2010 to 2020, take was mainly for protection of property (30% of take, mostly at landfills), public safety (26%, mostly at airports), and protection of threatened and endangered species (18%). Take assessments for the four gull species are needed to ensure that take levels do not negatively impact their populations.

Potential biological removal methods to estimate the maximum allowable annual take while keeping populations at sustainable levels were first developed for marine mammals (Wade et al. 1998). Runge et al. (2009) expanded the use of potential biological removal methods to include management of nuisance species and species open to sport harvest and changed the terminology to potential take level (PTL). Population size, intrinsic population growth rate, and a policy factor are the minimum information requirements for estimating PTL. The PTL approach is used to assess the lethal take of nongame migratory bird species, such as black vultures Coragyps atratus (Runge et al. 2009) and Nearctic songbirds (Johnson et al. 2012). To inform permitting decisions by the USFWS in the Atlantic Flyway, we used PTL methods to estimate an annual mortality limit for great black-backed, herring, laughing, and ring-billed gulls. Lethal management consists of shooting individual birds at problem areas and egg oiling or egg destruction at colony sites. Nonlethal methods of management are not considered in this assessment. Estimation of PTL provides a basis to evaluate proposed take levels at the Flyway scale. The PTL can be compared with desired take needed to achieve management goals and to refine management decisions.

Study area

We assessed the effect of lethal take on the four gull species in the U.S. portion of the Atlantic Flyway (Table 1). This area includes the islands and waters along the Atlantic Coast, and the Great Lakes in New York and Pennsylvania. The gull populations of concern breed and winter in the U.S. and Canadian portions of the Atlantic Flyway, which includes Labrador, Newfoundland, Quebec, and the provinces of the Canadian Maritimes. The laughing gull breeds along the U.S. Atlantic Coast and winters in the southern United States and into the Caribbean (Burger 2020). Herring and ring-billed gulls breed across much of Canada and winter throughout the southern United States (Pollet et al. 2020; Weseloh et al. 2020). Herring and ring-billed gulls that breed in the Great Lakes or interior Canada may winter in the U.S. portion of the Atlantic Flyway. The great black-backed gull breeds along the north Atlantic Coast of Canada south to the mid-Atlantic Coast of the United States (Good 2020). Great black-backed gulls winter offshore or in coastal areas, with most adults apparently remaining in northern waters while subadults migrate south to waters off the U.S. Atlantic Coast and most birds winter north of North Carolina (Good 2020).

Methods

PTL model

We need estimates of minimum population size (NMIN), one-half the intrinsic population growth rate (RMAX), and a recovery factor (FR) to calculate PTL:

$$PTL = N_{MIN} \cdot \frac{R_{MAX}}{2} \cdot F_R.$$

To guard against overexploitation due to uncertainty, we can use the number of individuals counted or the lower interval around a population estimate for NMIN (Runge et al. 2004). We based breeding population estimates for the four gull species on results from the 2013 Mid-Atlantic and New England Maritime Colonial Waterbird Survey and recent surveys conducted in other U.S. portions of the Atlantic Flyway (Table 1). These counts are uncorrected for detection probability or for breeding probability (i.e., counts were made at colony sites, yet not all birds of breeding age breed every year), thus the counts represent a minimum known number of breeding-age birds. We used a Leslie matrix approach (see below) to estimate the number of nonbreeders of each species (Table 1). We estimated NMIN as the sum of breeders plus nonbreeders assuming that take would come from all age classes (Dorr et al. 2016). If take is only of breeders at nesting colonies, the estimate of NMIN for calculating PTL should be reduced to include only breeding-age birds. Lethal control of the four species also includes take of nests (i.e., egg oiling and destruction). To incorporate the population-level effect of nest take, we assumed that oiling of a single nest removed 1.09 individuals (the product of hatch year survival [HY] [0.73] and number of young fledged per nest [1.5] for herring gulls). This estimate does not include the probability of renesting by individuals and assumes that oiling is 100% effective.

The value assigned to FR used to calculate PTL can range between 0.1 and 2.0 and reflects the policy
objective(s) for allowable take. When $F_R = 1$, PTL represents an estimate of maximum annual sustainable yield (Runge et al 2004) and results in a population at half of its theoretical carrying capacity. Setting $F_R < 1$ is appropriate if the objective is to manage for a smaller population. For marine mammals, Taylor et al. (2000) recommended $F_R = 0.1$ for endangered species and $F_R = 0.5$ for threatened species. We believed $F_R = 1.0$ would have been appropriate for each gull species if there was no concern for population status and if quality demographic data were available. Only the herring gull had complete demographic data, and a previous study (Niel and Lebreton 2005) estimated $R_{MAX}$ for this species. However, data from the all-bird Breeding Bird Survey for the Eastern Region suggested that the herring gull and great black-backed gull experienced long-term (1966–2017) declines in the northeast United States and Canada, while the ring-billed gull in the Great Lakes and laughing gull in the mid-Atlantic United States experienced population increases (Sauer et al. 2017; Pardieck et al. 2020). Therefore, we chose $F_R = 0.75$ for the herring (good data but declining), ring-billed, and laughing gulls (poor data but increasing) and $F_R = 0.50$ for the great black-backed gull (poor data and declining). To estimate if PTL for each species was reasonable, we compared PTL with average annual take between 2012 and 2019 (Table 2). Take data are from the Service Permit Issuance and Tracking System Database maintained by the U.S. Fish.

**Table 1.** Number of breeding and nonbreeding great black-backed gulls *Larus marinus*, herring gulls *L. argentatus*, ring-billed gulls *L. delawarensis*, and laughing gulls *Leucophaeus atericilla* in each state within the U.S. portion of the Atlantic Flyway 2013, unless otherwise noted. We took estimates of breeders from the literature and from personal communications, extrapolated the numbers of breeding individuals from counts of nests at colonies, and extrapolated the number of nonbreeding individuals from counts of breeding individuals using a Leslie matrix, assuming stable age populations. We used total abundance estimates ($N_{MIN}$) in estimating annual mortality limits.

| Breeders | Great black-backed gull | Herring gull | Laughing gull | Ring-billed gull |
|----------|------------------------|--------------|---------------|-----------------|
| Connecticut* | 358 | 1,178 | 0 | 0 |
| Delaware* | 0 | 0 | 18,000 | 0 |
| Florida | 0 | 0 | 49,000 | 0 |
| Georgia | 0 | 0 | 236 | 0 |
| Maine** | 13,868 | 42,976 | 6,354 | 0 |
| Maryland | 842 | 3,612 | 1,898 | 0 |
| Massachusetts* | 9,038 | 15,008 | 3,726 | 0 |
| New Hampshire | 0 | 0 | 0 | 0 |
| New Jersey | 2,931 | 4,206 | 36,926 | 0 |
| New York*** | 6,982 | 10,804 | 4,802 | 123,272 |
| North Carolina | 0 | 0 | 8,840 | 0 |
| Pennsylvania** | 445 | 14,990 | 12,050 | 5,743 |
| Rhode Island | 1,838 | 3,866 | 0 | 0 |
| South Carolina | 0 | 0 | 13,126 | 0 |
| Vermont | 24 | 480 | 0 | 27,156 |
| Virginia | 2,238 | 3,914 | 33,306 | 0 |
| West Virginia | 0 | 0 | 0 | 0 |
| U.S. Flyway Total | 38,564 | 101,034 | 188,264 | 156,171 |

| Nonbreeders | Great black-backed gull | Herring gull | Laughing gull | Ring-billed gull |
|-------------|------------------------|--------------|---------------|-----------------|
| 1 yo | 10,582 | 28,290 | 27,411 | 22,738 |
| 2 yo | 7,619 | 20,368 | 19,736 | 16,372 |
| 3 yo | 6,629 | 17,721 | — | — |
| 4 yo | 5,767 | 15,417 | — | — |
| $\approx N_{MIN}$ | 70,000 | 180,000 | 240,000 | 200,000 |

yo = years old.
* Mid-Atlantic and New England Maritime Colonial Waterbird Survey, 2013.
** Craig Faulhaber, Florida Fish and Wildlife Conservation Commission.
*** Tim Keyes, Georgia Department of Natural Resources.
**** USFWS Final Environmental Assessment: Laughing Gull Management Plan for Seabird Restoration Islands in Maine, 2008.
**ỳ J. Ozard, New York Department of Environmental Conservation (personal communication), data for ring-billed, great black-backed, and herring gulls.
† USDA Wildlife Services Environmental Assessment 2018 for North Carolina.
‡ Pennsylvania numbers include 2,689 ring-billed gull rooftop nests ($^2 = 5,378$ adults) removed by the USDA in Erie, Pennsylvania, during 2016 (Matthew Rice, USDA Wildlife Services). All other numbers in Pennsylvania are based on a May 2013 survey of nonbreeding gulls in Morrisville, Pennsylvania, near a landfill (Jason Wood, USDA Wildlife Services).
§ USDA Wildlife Service Environmental Assessment for South Carolina, and Felicia Sanders, South Carolina Department of Natural Resources.
‡ USDA Wildlife Services annual depredation permit report: gull control activities on state-owned Islands, Lake Champlain, Vermont, 2013 (Fred Pogmore, USDA Wildlife Services). Includes New York Islands within Lake Champlain.
¶ Watts et al. 2019.
Table 2. Annual permitted lethal take from 2012 to 2019 of great black-backed gulls Larus marinus, herring gulls L. argentatus, ring-billed gulls L. delawarensis, and laughing gulls Leucophaeus atricilla from the Atlantic Flyway, United States. Take is divided between the killing of individuals (Individuals) or the oiling or destruction of nests with eggs (Nests).

| Year | Individuals | Nests | Individuals | Nests | Individuals | Nests | Individuals | Nests |
|------|-------------|-------|-------------|-------|-------------|-------|-------------|-------|
| 2012 | 1,104       | 798   | 8,002       | 3,241 | 6,393       | 4,596 | 4,229       | 1,163 |
| 2013 | 951         | 1,343 | 8,213       | 6,064 | 5,695       | 4,627 | 4,925       | 5,014 |
| 2014 | 598         | 733   | 6,981       | 2,839 | 5,912       | 7,018 | 5,512       | 7,687 |
| 2015 | 552         | 712   | 6,755       | 4,334 | 4,937       | 6,038 | 4,260       | 8,646 |
| 2016 | 370         | 751   | 4,519       | 2,964 | 4,749       | 6,939 | 4,092       | 5,514 |
| 2017 | 477         | 305   | 5,871       | 2,075 | 5,957       | 3,970 | 4,218       | 4,383 |
| 2018 | 538         | 442   | 5,636       | 3,744 | 4,242       | 6,676 | 5,058       | 5,146 |
| 2019 | 328         | 86    | 4,704       | 1,743 | 4,992       | 1,181 | 3,807       | 2,642 |

For the life table approach, we used eigen analysis of population projection matrices to estimate \( R_{MAX}^{MM} \) (Leslie 1945, 1948; Lefkovitch 1965; Caswell 2001). The matrix model (MM) accommodates age structure in a discrete time model with age cohorts, which we can express as a projection matrix (Caswell 2001). For example, the herring gull does not breed until 5 years old then produces an average number of young per breeding age individual \( b \), has first \( s_1 \) and second year \( s_2 \) survival rates that are lower than after SY survival, and survival for individuals in their third year and older is the same \( s_3 = s_4 = s_5 \). Abundance from one year \( N_t \) to the next is expressed via matrix multiplication:

\[
\begin{bmatrix}
N_{1,t} \\
N_{2,t} \\
N_{3,t} \\
N_{4,t} \\
N_{5,t}
\end{bmatrix} =
\begin{bmatrix}
0 & 0 & 0 & 0 & bs_1 \\
0 & s_2 & 0 & 0 & 0 \\
0 & 0 & s_3 & 0 & 0 \\
0 & 0 & 0 & s_4 & 0 \\
0 & 0 & 0 & 0 & s_5
\end{bmatrix}
\begin{bmatrix}
N_{1,t-1} \\
N_{2,t-1} \\
N_{3,t-1} \\
N_{4,t-1} \\
N_{5,t-1}
\end{bmatrix}
\]

The dominant eigenvalue of the matrix is \( \lambda_{MAX}^{DIM} \). Optimum generation time \( T_{OPT} \) is calculated as:

\[
T_{OPT} = \frac{\lambda_{MAX}^{DIM} - s}{s} \]

where \( s \) is adult survival. We obtained population vital rate estimates (Table 3) from the literature for each gull species or used surrogate estimates from closely related species. We used a simulation approach with 10,000 draws to estimate \( R_{MAX}^{DIM} \) and we randomly selected parameter estimates from either uniform or normal distributions based on the range of values or standard errors (Table 3). Simulations were performed in program R (R Core Team 2020).

Because the life table approach requires using demographic estimates that are uncertain, and individual population dynamics usually vary from estimates produced by the allometric approach, Dillingham et al. (2016) developed a method combining the two approaches they termed the \( \sigma T \)-adjusted method. First, they assumed \( R_{MAX}^{DIM} T_{OPT} = a_\sigma \alpha T = a_\sigma \sigma T \) and that variability in fit of the allometric model can be accommodated as:

\[
R_{MAX} T_{OPT} = (a_\sigma \sigma T)
\]
Here, $a_{\alpha T}$ ($a_{\alpha T} = a_{\alpha} a_{T}$) is the allometric constant, and $\sigma_{\alpha T}$ is the population-level standard deviation. We used the allometric constant estimated by Niel and Lebreton (2005) for 13 species of birds ($a_{\alpha T} = 1.0633$, 95% confidence interval [95% CI] = 0.9783–1.1483) for simulating $a_{\alpha T}$ values using the DIM approach. We then combined simulated values of $R_{\text{MAX}}^{\text{DIM}}$ and $T_{\text{OPT}}^{\text{DIM}}$ with the allometric model by keeping only estimates that were within a tolerance of $\delta < 0.05$ (i.e., $|R_{\text{MAX}}^{\text{DIM}} T_{\text{OPT}}^{\text{DIM}} - R_{\text{MAX}}^{\text{OPT}} T_{\text{OPT}}^{\text{OPT}}| < \delta$). We derived these combined estimates of intrinsic growth and generation time ($R_{\text{MAX}}^{\text{OPT}}$, $T_{\text{OPT}}^{\text{OPT}}$) from the intervals of the simulation results. Combining the two methods using this approach relaxes the constraint that $a_{\alpha T} = 1$ but keeps the $R_{\text{MAX}}$ estimate near what is expected from the allometric model (Dillingham et al. 2016). We used $R_{\text{MAX}}$ for calculating PTL for each gull species. For inference, we derived adult survival under optimal conditions ($s_{\text{OPT}}$) with a stated numerical tolerance $\delta$ by only using the survival estimate from MM simulations that agreed with the allometric constant $a_{\alpha T}$. We derived $s_{\text{OPT}}$ because it is difficult to know if studies used to estimate $R_{\text{MAX}}^{\text{DIM}}$ coincided with periods of $R_{\text{MAX}}^{\text{OPT}}$.

**Demographic data**

**Herring gull.** Individuals typically begin breeding at 5 years old but can first breed as early as 4 and as late as 6 years of age (Pierotti and Good 1994; Weseloh et al. 2020). Age at last breeding is unknown but is believed to be around 20 years old, and maximum observed lifespan is 30 years old. We estimated the number of female offspring per female ($b$; Table 3) from raw numbers presented by Pierotti and Good (1994); of 660 individuals, 23% fledged three young, 20–30% fledged two young, 20–30% fledged on young, and 15–30% fledged zero young. This fledge rate was similar to that ($b = 0.765$) reported by Mineau and Weseloh (1981) for eight colonies over 3 y in the Great Lakes. We used estimates for HY and SY annual survival (Table 3) from a European study of herring gulls (Lebreton et al. 1995). Two studies that relied on capture-recapture methods estimated adult (after SY [ASY]) survival at 0.87 (Allard et al. 2006) and 0.91 (Breton et al. 2008). European studies suggest that ASY survival may be higher, around 0.93 or 0.94 (Chabrzyk and Coulson 1976; Lebreton et al. 1995). To encompass the range of estimates, we simulated data from a uniform range of 0.87 to 0.94. Some herring gulls originating from northeast Quebec, Newfoundland, and the Great Lakes migrate through or winter in the U.S. portion of the Atlantic Flyway (Moore 1976; Threlfall 1978). We did not count individuals from these areas to estimate $N_{\text{MIN}}$ because their exact number was unknown and whether they would be exposed to take.

**Great black-backed gull.** We used survival estimates from the herring gull because no survival estimates were available for the great black-backed gull (Table 3). Maximum observed longevity is 19 years old, and great black-backed gulls begin breeding in their fourth or fifth year of life (Good 2020). Good (2020) reported the percent of breeders fledging three, two, one, and zero young on Appledore Island, Maine, for 1980 and 1995. Using these percentages and the numbers of great black-backed gulls on Appledore Island (MANEM 2006), $b$ was 0.784 (SE = 0.018) for 1980 and 0.440 (SE = 0.011) for 1995. We used the 1980 estimate of $b$ for calculations because the population was much lower in 1980. Great black-backed gulls that breed along the northern Atlantic Coast in Canada may winter in the area where take may occur, whereas most breeding individuals south of Maine likely remain near their breeding areas year round (Good 2020). We did not count individuals that breed in Canada to estimate $N_{\text{MIN}}$ because their exact number was unknown or whether they would be exposed to take.

**Laughing gull.** We found no survival estimates for any age class for the laughing gull. In a study of black-headed gulls *Chroicocephalus ridibundus* (a similarly sized “hooded” gull) in France, Prévot-Julliard et al. (1998) estimated adult survival to be 0.90. Prévot-Julliard et al. (1998) thought this estimate was high relative to estimates from other gull species but believed it was accurate because the colony under study was stable and productive. We used HY and SY survival estimates from the herring gull, although they may be high because herring gulls are larger than laughing gulls. Maximum known longevity is 19 years old, and age of first breeding is 2 or 3 years old (Burger 2020). Sparse estimates of the number of young fledged per nest or per pair were

**Table 3.** Distributions of demographic parameters used for estimating maximum population growth rate and mortality limits for great black-backed gulls *Larus marinus*, herring gulls *L. argentatus*, ring-billed gulls *L. delawarensis*, and laughing gulls *Leucophaeus aterricilla* in the Atlantic Flyway, United States. We took values for demographic parameters from the literature, where $s$ is annual survival, $b$ is the number of female offspring per female, and $\alpha$ is age at first reproduction. Distributions are normal (mean ± SD) or uniform. We took distributions from studies published since 1977 and took demographic parameters and their uncertainty from studies published from 1974 to 2020.

| Parameter | Age-class | Great black-backed gull | Herring gull | Laughing gull | Ring-billed gull |
|-----------|-----------|-------------------------|-------------|---------------|-----------------|
| $s$       | HY        | 0.729 ± 0.035           | 0.729 ± 0.035 | 0.729 ± 0.035 | 0.729 ± 0.035   |
| $s$       | SY        | 0.886 ± 0.024           | 0.886 ± 0.024 | 0.886 ± 0.024 | 0.886 ± 0.024   |
| $s$       | ASY       | U (0.870–0.940)         | U (0.870–0.940) | U (0.86–0.92) | U (0.86–0.92)   |
| $b$       |           | 0.784 ± 0.018           | 0.752 ± 0.022 | U (0.6–0.7)   | U (0.85–1.10)   |
| $\alpha$  |           | U (4–5)                 | U (4–6)      | U (2–3)       | U (2–4)         |

ASY = all years after second year; HY = hatch year; SY = second year; U = uniform.
available for the laughing gull and ranged from 0 to 1.4 (see appendix in Burger [2020]). Laughing gull clutch size is 2.5 to 2.8, and hatching success is 70–80% (Dinsmore and Schreiber 1974; Burger 2020). The studies summarized in Burger (2020) that reported fledglings per pair fell into two groups: five studies in the 0 to 0.7 range and five studies in the 1.2 to 1.4 range. We consider this latter range of values as coming from populations growing at or closer to $R_{\text{MAX}}$. To simulate $b$, we used a range of 0.6 to 0.7.

Ring-billed gull. Age at first breeding is largely unknown but appears to be 2 to 4 years old (Ludwig 1974; Southern and Southern 1980; Pollet et al. 2020). Ludwig (1974) estimated 1.74 fledglings per pair in rapidly growing colonies in Lakes Huron and Michigan. The number of fledglings per pair from eight studies of 14 colonies (reported in Pollet et al. [2020]) ranged from 0.6 to 2.2. There was some general clustering of values around 0.6, 1.0, and 1.7 to 2.2. To simulate $b$, we used a uniform distribution of 0.85–1.10 (Table 3). Annual survival is largely unknown. Ludwig (1974) and Southern (1977) believed adult annual mortality was probably 12–13%. We thought 0.87 would be low relative to the other three species assessed herein, and we used the laughing gull estimate for ASY survival (Table 3). We used the herring gull estimates of HY and SY survival for simulations. Ring-billed gulls do not nest along the U.S. Atlantic Coast but nest in the Great Lakes portion of the Atlantic Flyway, Lake Champlain, and farther north in Newfoundland and northeastern Quebec. Based on data from the U.S. Geological Survey Bird Banding Lab, ring-billed gulls banded in the Great Lakes are recovered along the Atlantic Coast, but it was not clear if individuals from Newfoundland and northern Quebec would be exposed to take in the United States because few individuals have been banded in this region. To be conservative, we assumed that all harvest would be from individuals that originated in the U.S. portion of the Atlantic Flyway in the Great Lakes (Table 1).

Results

Herring gull

We estimated a minimum population size of 180,000 in the Atlantic Flyway (Table 1). The $R_{\text{MAX}}$ estimate using the DIM approach ($b_{\text{DIM}} = 0.114$, 95% CI = 0.095–0.135) was similar to that of the life table approach ($R_{\text{MM}} = 0.120$, 95% CI = 0.083–0.163). Estimated generation time under optimal conditions was 9.067 (95% CI = 7.756–10.633), and $a_{\text{IT}}$ was 1.084 (95% CI = 0.777–1.400). The combined estimate of $R_{\text{MAX}}$ was closer to the life table approach ($R_{\text{MM}} = 0.114, 95\% \text{ CI} = 0.096–0.147$), but confidence intervals were overlapping for all three estimates. The estimate for optimum survival was lower than that used for the life table approach but within bounds of uncertainty ($s_{\text{OPT}} = 0.890$, 95% CI = 0.885–0.920). Simulated $R_{\text{MAX}}$ and $s_{\text{OPT}}$ results from the MM varied widely from the allometric approach (Figure 1), with only 24% within the tolerance limit. This was because more parameters with accompanying uncertainty were required for the matrix approach, and the estimates were not clumped near the intersection of the two curves (Figure 1). The same relationship (more variable estimates using the MM approach) held for the other species. Average annual take of herring gulls in the Atlantic Flyway during 2012–2019 was 10,134 individuals (SD = 2,617), primarily to protect human health and safety (municipalities and airports) and to protect state-threatened and/or federally threatened or endangered species from gull predation. Allowable annual take (PTL) is 7,963 (95% CI = 6,553–9,871). There was 94% overlap between the distributions of observed and allowable take, and 79% of the observed take distribution was >7,963 (Figure 2).

Great black-backed gull

We estimated a minimum population of 70,000 individuals in the Atlantic Flyway (Table 1). The $R_{\text{MAX}}$ estimate using the DIM approach ($b_{\text{DIM}} = 0.114, 95\% \text{ CI} =$
0.095–0.135) was lower than using the life table approach ($R_{\text{MM}}^{\text{MAX}} = 0.124$, 95% CI = 0.088–0.168). Estimated generation time under optimal conditions was 8.974 (95% CI = 8.674–9.315), and $\alpha_T$ was 1.119 (95% CI = 0.817–1.411). The combined estimate of $R_{\text{MAX}}$ was in between the DIM and life table approaches, but confidence intervals were overlapping ($R_{\text{MAX}}^{\text{DIM}} = 0.119$, 95% CI = 0.108–0.130). The estimate for optimum survival was lower than that used for the MM but within bounds of uncertainty ($s_{\text{OPT}}^{\text{DIM}} = 0.898$, 95% CI = 0.882–0.914). Average annual take of great black-backed gulls in the Atlantic Flyway during 2012–2019 was 1,319 (SD = 633) individuals. Allowable annual take is 2,081 (95% CI = 1,881–2,280). There was 37% overlap between the distributions of observed and allowable take, and 11% of the observed take distribution was >2,081 (Figure 2).

Laughing gull

We estimated a minimum population of 240,000 individuals in the Atlantic Flyway (Table 1). The $R_{\text{MAX}}$
estimate using the DIM approach ($R^\text{DIM}_{\text{MAX}} = 0.174, 95\% \text{ CI} = 0.151–0.210$) was higher than that using the life table approach ($R^\text{MM}_{\text{MAX}} = 0.160, 95\% \text{ CI} = 0.109–0.208$). Estimated generation time under optimal conditions was 6.237 (95\% CI = 5.916–6.610), and the $s_{\text{F}}$ was 0.953 (95\% CI = 0.755–1.141). The combined approach suggested that $R^\text{MAX}$ was in between the DIM and life table approaches, but confidence intervals were overlapping ($R^\text{MM}_{\text{MAX}} = 0.167, 95\% \text{ CI} = 0.150–0.184$). The estimate for optimum survival was lower than that used for the matrix approach but within bounds of uncertainty ($s^\text{OPT}_{\text{DIM}} = 0.907, 95\% \text{ CI} = 0.884–0.919$). Average annual take of laughing gulls in the Atlantic Flyway during 2012–2019 was 10,952 (SD = 2,966) individuals. Allowable annual take is 15,039 (95\% CI = 13,514–16,565). There was 29\% overlap between the distributions of observed and allowable take, and 3\% of the observed take distribution was >15,039 (Figure 2).

**Ring-billed gull**

We estimated a minimum population of 200,000 individuals in the Atlantic Flyway (Table 1). The $R^\text{MAX}$ estimate using the DIM approach ($R^\text{DIM}_{\text{MAX}} = 0.174, 95\% \text{ CI} = 0.151–0.210$) was lower than using the life table approach ($R^\text{MM}_{\text{MAX}} = 0.222, 95\% \text{ CI} = 0.162–0.313$). Estimated generation time under optimal conditions was 5.482 (95\% CI = 5.209–5.817), and the $s_{\text{F}}$ was 1.218 (95\% CI = 1.022–1.401). The combined estimate of $R^\text{MAX}$ was closer to the estimate from the life table approach, but confidence intervals were overlapping ($R^\text{MM}_{\text{MAX}} = 0.197, 95\% \text{ CI} = 0.177–0.220$). The estimate for optimum survival was lower than that used in the MM but within bounds of uncertainty ($s^\text{OPT}_{\text{DIM}} = 0.871, 95\% \text{ CI} = 0.861–0.896$). Average annual take of ring-billed gulls in the Atlantic Flyway during 2012–2019 was 9,989 (SD = 2,966) individuals. Allowable annual take is 14,826 (95\% CI = 13,310–16,476). There was 49\% overlap between the distributions of observed and allowable take, and 19\% of the observed take distribution was >14,826 (Figure 2).

**Discussion**

The methods we used for estimating $R^\text{MAX}$ (Dillingham et al. 2016) were useful, but estimates from the demographic-invariant and life table approaches were similar. The similarity was likely because the key demographic parameters (age at first breeding and adult survival) were close to what occurs when gull populations are growing at $R^\text{MAX}$. The usefulness in combining the DIM and MM approaches to estimate $R^\text{MAX}$ should be apparent when there is little demographic information. For example, not having estimates of HY and SY survival for herring gulls would have required more assumptions when using the MM for all species. In the complete absence of demographic information, it is still possible to use body size to approximate demographic parameters (Fenchel 1974; Blueweiss et al. 1978).

We assumed that the demographic parameter estimates we used were representative of the four gull species at $R^\text{MAX}$, but only the herring gull had multiple estimates of each demographic parameter, and only the herring gull had estimates where authors noted that the population or populations under study were at low density with no known constraints. Our $R^\text{MAX}$ estimates were within expected values for species in the family Laridae (Niel and Lebreton 2005). The DIM and life table approaches produced estimates of $R^\text{MAX}$ that were greater for the smaller two species (laughing and ring-billed gulls) than the larger two species (great black-backed and herring gulls), and this agreed with hypothesized allometric relationships of body mass and intrinsic population growth rate (Blueweiss et al. 1978). Although the estimates of $R^\text{MAX}$ from the two approaches had overlapping confidence intervals for each species, estimates of $R^\text{MAX}$ from the DIM approach were consistently lower, except for laughing gulls. We suspect the fecundity estimate used for the laughing gull was relative to what occurs under optimal environmental conditions. Using HY and ASY survival from the herring gull for all species may have biased $R^\text{MAX}$ because the herring gull is larger than the ring-billed and laughing gulls and may have higher survival rates (Blueweiss et al 1978). However, we suspect that adult survival under conditions of $R^\text{MAX}$ varies little among Laridae species. Other authors have also hypothesized this to be the case (Ludwig 1974; Niel and Lebreton 2005).

We assumed that nest counts (Table 1) were representative of current population abundance. Population size estimates were likely underestimates because detection probability of individuals in colonies and of colonies on the landscape was not accounted for, and because we assumed all birds of breeding age were nesting. Underestimates of population size resulted in conservative estimates of PTL for each species. In addition, we reduced PTL for herring and great black-backed gulls because their populations likely declined subsequent to the counts used to estimate $N_{\text{MIN}}$. We suggest using a conservative approach to estimate PTL until managers collectively determine population objectives for the four gull species within the Atlantic Flyway. If the policy objective becomes to manage for smaller population sizes, increasing the values used for $F_R$ in the PTL calculations would be appropriate, keeping in mind that the distribution and population status of each species is different. The herring gull has highly variable migratory patterns, but herring gulls that breed along the north Atlantic Coast of the United States and in the Great Lakes apparently winter in their breeding areas. Herring gulls that breed further north in Canada probably migrate further south in the United States (Weseloh et al. 2020). In addition, there is moderate conservation concern for herring gulls around the Great Lakes and Atlantic Coast in Canada (Environment Canada 2019). Although we took a conservative approach to estimating $N_{\text{MIN}}$ for all species, some herring gulls from Canada will be exposed to take in the United States. This is also true of ring-billed gulls. We recommend that setting policy objectives for herring and ring-billed gull population sizes be done in consultation with Environment and Climate Change Canada to guard against excessive take by the two countries. We also
recommend that allowable take be coordinated between the two federal (USFWS) regions in the Atlantic Flyway for all four species.

Our assessments for great black-backed and laughing gulls suggested that expected annual take, based on average annual take from 2012 to 2019, will be below the PTL. The expected annual take for ring-billed gulls will exceed PTL about 20% of the time, whereas expected annual take for herring gulls will exceed PTL about 80% of the time. Annual take occasionally exceeding PTL might not be of concern if the excess is slight and does not occur in consecutive years; however, there is only partial controllability in realized annual take. For example, on average, 31,400 individual herring gulls were authorized for take each year between 2012 and 2019, but an annual average of 6,700 were reported killed (numbers do not include authorized or actual take of nests with eggs). Permit requests may far exceed take because managers often need to project annual take 3 y in advance for permit requests. If take above PTL does occur, we recommend that authorized take in subsequent years be reduced to compensate and the population(s) be monitored. Managers can factor in the historic ratio of authorized to actual take, particularly for permit renewal applications, when determining how many permits to issue or the amount of lethal take to authorize on individual permits. If lethal take becomes restricted, the authorized-to-actual take ratios may decrease if applicants and managers become more selective formulating permit requests and subsequent authorizations. Prioritizing permits for issues of health and human safety may also occur.

Allowable maximum take level for each species should depend on the objectives for population and conflict management. The objective we sought to address was to ensure conservation of each gull species by only allowing take at sustainable levels. We do not infer that take should be at maximum allowable every year. If this were to occur, we recommend re-evaluating this assessment at frequent intervals, such as every 5 y, to ensure that take is sustainable and that the dynamics of the gull populations have not changed. We estimated take capacity for the gull species at the spatial scale of the U.S. portion of the Atlantic Flyway and did not consider the possibility of declines or extirpation at local scales, such as in individual states or at individual breeding colonies. This may be a concern to local managers. In addition, we assumed that take of each species would come from all age classes. If take is only at colony sites and only breeding birds are killed, PTL should be recalculated using a smaller population size. For reference, in the U.S. portion of the Atlantic Flyway, approximately 30–40% of annual take of herring and laughing gulls was from nest or egg destruction, whereas almost no ring-billed or great black-backed gull nests or eggs were destroyed.

Lethal take is intended to help reduce human–bird conflicts to safe and practical levels. At one extreme, managers will use lethal take to reduce avian-caused human mortality from catastrophic airplane birdstrikes. At the other extreme, the four gull species may be perceived as a nuisance, and management actions other than lethal take, such as hazing, may be effective. The USFWS requires applicants to list nonlethal hazing and habitat management deterrents that have been tried when applying for a permit for lethal take, and lethal take is a short-term solution until long-term nonlethal measures are in place. It will fall to regional and local managers to best manage human–bird conflicts by balancing the use of nonlethal techniques with the permitting of lethal take.

Although the need for continued monitoring of gull abundance and take may seem obvious, our experience suggests that monitoring programs are difficult to maintain. A population-monitoring program can reduce uncertainty in $N_{\text{MIN}}$ and $R_{\text{MAX}}$, reducing the risk of overexploitation if severe population declines can be quickly detected and lethal take adjusted. Continued monitoring is also necessary to evaluate the assumptions underlying the PTL and to adjust the recovery factor if gull–human conflicts increase or decrease. The USFWS is legally mandated to manage migratory bird populations and develop and enforce regulations that govern the take of migratory birds (Migratory Bird Treaty Act). To meet this legal mandate, we believe that the permitted killing of gulls or any species that will occur for multiple years, or where take will exceed established limits, needs to be accompanied by monitoring to assess changes in population status.

Supplemental Material

Please note: The Journal of Fish and Wildlife Management is not responsible for the content or functionality of any supplemental material. Queries should be directed to the corresponding author for the article.

Reference S1. Dolbeer RA, Begier MJ, Miller PR, Weller JR, Anderson AL. 2021. Wildlife strikes to civil aircraft in the United States 1990–2019. Washington, D.C.: Federal Aviation Administration National Wildlife Strike Database Report Number 26, Report to the Associate Administrator of Airports Office of Airport Safety and Standards Airport Safety & Certification. Available: https://doi.org/10.3996/JFWM-20-088.S1 (3.5 MB PDF)

Reference S2. [MANEM] Mid-Atlantic/New England/ Maritimes Region Waterbird Working Group. 2006. Waterbird conservation plan: 2006–2010 Mid-Atlantic/ New England/Maritimes Region. A plan for the Waterbird Conservation for the Americas Initiative. Available: https://doi.org/10.3996/JFWM-20-088.S2 (1.33 MB PDF)

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