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Population Management to Reduce the Risk of WiId life-Aircraft Collisions

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Four basic control strategies mitigate the risks to aviation caused by wildlife at airports: (1) aircraft flight schedule modification (primarily at military airbases) and enhancement of aircraft visibility to avoid interactions with wildlife (e.g., Blackwell et al. 2009*b*, 2012); (2) habitat modification and elimination of food, water, and cover that attract wildlife (Cleary and Dolbeer 2005, Blackwell et al. 2009*a*; Chapters 5, 8–10); (3) repellent and harassment techniques to disperse wildlife (Cleary and Dolbeer 2005; Chapters 2–4); and (4) wildlife population management (e.g., Dolbeer 1998). As discussed throughout this book, successful efforts to mitigate the risk of wildlife–aircraft strikes at airports usually involve programs that attempt to integrate these strategies. This chapter focuses on wildlife population management.

In general, wildlife population reduction by killing or through reproductive control at or in the vicinity of an airport is the last option deployed after all other actions have been considered or implemented. However, management of a wildlife hazard situation at an airport may require killing an individual animal, or require that a local population of a problem species be reduced by lethal or reproductive means until, if feasible, a long-term, nonlethal solution can be implemented (e.g., erecting a deer-proof fence, relocating a nearby gull [Laridae] nesting colony; see Chapters 5–6). In addition, lethal removal of a few individuals sometimes reinforces nonlethal frightening techniques (Baxter and Allan 2008). Recurrent lethal control is often necessary as part of an integrated Wildlife Hazard Management Plan (WHMP) for an airport (Cleary and Dolbeer 2005, Baxter 2008).

Most wildlife species that frequent airport environments are protected by some combination of federal, state, and local laws, often requiring permits before any action can be taken to capture or kill animals or to control their reproduction. Ninety percent of the birds struck by civil aircraft in the USA are species federally protected by the Migratory Bird Treaty Act (Dolbeer et al. 2012). Permits require justification of why the removal is needed, the numbers to be removed by species, and the methods used to remove and dispose of the animals. In addition, management of wildlife populations often generates public interest, which airports must acknowledge and address. The following steps should be taken to justify population reduction through lethal or reproductive control and to minimize adverse public reaction to a program involving killing wildlife:

- Document that the wildlife species is an economic, safety, or health threat.
- Justify why nonlethal options alone are not adequate to solve the problem.
- Assess the impact that the lethal or reproductive control will have on local and regional populations of the species (i.e., is the action likely to result in a significant reduction in numbers of the species at the local or regional level?).
- Assure that the methods are appropriate (i.e., legal, safe, effective, and humane) and specific for the targeted wildlife species.

- Document the number of animals killed or treated by species.
- Document the effectiveness of population management actions in mitigating the problem (e.g., reduction in numbers observed at airports and in wildlife strikes).
- Recommend steps to be taken, if any are feasible, to reduce the need for population management actions in the future.
- Issue timely reports, preferably annually, that summarize the items listed above. Transparency increases public acceptance and allows for more effective adaptive management strategies.

Three critical types of information are needed for airports to justify lethal or reproductive control programs to regulatory agencies and the public before implementing these programs. First, the hazard level and the risk posed by the wildlife species must be documented (Dolbeer and Wright 2009). Lethal control may be warranted at a particular airport for species such as Canada geese (*Branta canadensis*) or white-tailed deer (*Odocoileus virginianus*) that have a high hazard level (i.e., >50% of strikes with aircraft result in damage; Dolbeer et al. 2012) and that pose a high risk (i.e., the species have been documented through observations or strike events to frequent the airport; see also Biondi et al. 2011, DeVault et al. 2011). In contrast, at the same airport it may be inappropriate to request a permit for lethal control for a species such as American kestrel (*Falco sparverius*) with a relatively low hazard level (<2% of strikes cause damage) and that is infrequently observed.

An understanding of the local and regional population status and dynamics of the problem species is also needed before developing a management plan. Population data from local surveys, breeding bird surveys, Christmas bird counts, and other sources can be integrated with reproductive and survival rates to develop simple population models for the species of concern (Dolbeer 1998, Runge et al. 2009). These models can predict the immediate impact that lethal or reproductive control programs will have on local or regional populations and project how populations will respond to these management actions (e.g., Blackwell et al. 2003, Runge et al. 2009). Such models provide a scientific foundation to guide management actions and to provide a level of objectivity in the emotional debates that often arise when proposals are made to kill or reduce reproductive rates of wildlife (Dolbeer 1998).

Finally, airports must monitor the population level of the targeted species, as well as the number of strikes and associated damage (DeVault et al. 2011) caused by that species before and after implementing the population management plan. Monitoring allows for documentation of the effects that management actions have on the population and, most importantly, on the number of strikes.

These three types of information would be ideally integrated into regional strategic plans that encompass all airports within a specified area, allowing for more efficient permitting, implementation, and monitoring of target wildlife species. An emphasis on regional, rather than national, strategies takes into account that problem wildlife species in one area may not necessarily be problems in another area. In addition, the incorporation of adaptive management into regional strategic plans would allow for more efficient "learning while doing." Adaptive management is a formal, structured process that allows for flexible decision making in the face of uncertain outcomes from management practices and natural variability (Williams et al. 2007). Successfully used in regional management of natural resources (e.g., Weinstein et al. 1996), this approach has direct application to management of wildlife populations at airports.

Primer of Population Dynamics

Any consideration of management of wildlife populations by airport biologists, particularly lethal management, should be grounded in a basic understanding of wildlife population dynamics from spatial and temporal perspectives. In particular, effects of population demography (age and sex ratios, reproductive rates) and seasonal habitat and foraging requirements will influence how populations use airport environments. For most widely distributed wildlife populations, airports represent relatively small management units that may be used differently depending on the season. As such, wildlife habitats within airport perimeters probably do not sustain distinct population segments, but environments outside airport perimeters bolster these populations. Airports generally represent microcosms

within a larger landscape, and effective management of wildlife within these microcosms depends on the species, characteristics of their population dynamics, their habitats, and other spatial and temporal factors affecting their populations.

Wildlife populations occur at a variety of spatial scales, ranging from small, isolated populations to continent-wide populations. These populations can also vary temporally (e.g., daily, seasonally, or annually) at any given location. Rates of population growth (λ) for wildlife populations depend on several species-specific demographic components, such as annual survival, reproductive output, immigration, and emigration. In terms of measurable quantities, λ can be expressed as:

$$\lambda = \phi + f,$$

where ϕ is apparent survival for older age classes (a function of survival and emigration) and f is recruitment (a function of reproductive output and immigration; Nichols et al. 2000). In turn, these demographic characteristics are dependent on spatial factors such as habitat suitability and quality, as well as temporal factors such as seasonal weather conditions. Under the concept of *r* and *K* selection (Stearns 1976, Boyce 1984), there exists a continuum of life history strategies relevant to population dynamics, where *r*-selected species mature early and have high reproductive output and low adult survival (low ϕ and high f in the above equation), and where *K*-selected species mature late and have low reproductive output and high adult survival (high ϕ and low f in the above equation). This range of different life-history strategies will affect the success of methods used to manage populations (see below).

One key, underlying factor controlling population dynamics is habitat (e.g., Pulliam and Danielson 1991). Habitat is a species-specific concept; each species has unique habitat requirements. In terms of populations, habitat quality is a key concept and can be defined as the "ability of the environment to provide conditions appropriate for individual and population persistence" (Hall et al. 1997). Habitat quality is linked inextricably with population performance from small to large scales. Habitat quality governs larger-scale metapopulation processes such as source–sink dynamics, where population sources may reside in areas of high-quality habitat that then contribute individuals to areas of low-quality habitat (sinks) through recruitment (primarily immigration; Pulliam 1988, Pulliam and Danielson 1991, Runge et al. 2006; but see also Doncaster et al. 1997).

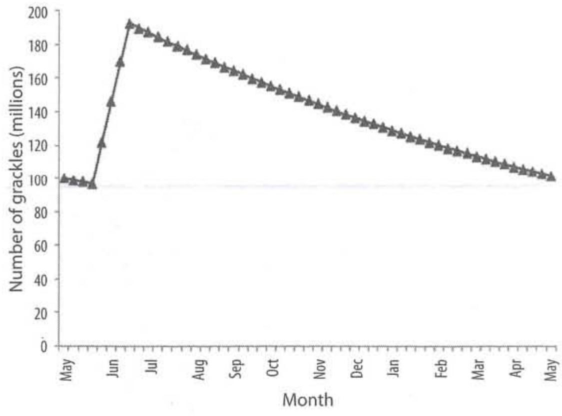

Fig. 7.1. Simulated annual cycle of the common grackle population in eastern North America, demonstrating the dynamic nature of wildlife populations. Adapted from Dolbeer (1998)

Also, unlike human populations, wildlife populations often exhibit dramatic within-year (annual) cycles in numbers. Most wildlife species have a narrow season of births followed by fledging/weaning, which introduces a large pulse of young animals into the population each year. This pulse of young animals occurs in summer for most species at the middle to high latitudes typical of Europe and North America. The magnitude of the annual population cycle is related to the age-specific reproductive rate of the species. Species such as snowshoe hares (*Lepus americanus*; Dolbeer and Clarke 1975) and red-winged blackbirds (*Agelaius phoeniceus*; Dolbeer et al. 1976) have pronounced annual cycles because females are sexually mature at one year old and are capable of producing several young each year. The common grackle (*Quiscalus quiscula*, a species with a similar life history as the red-winged blackbird) population in the eastern USA is estimated to be about 100 million at the start of the nesting season in April. By June, when young have fledged (a mean of about two per female one year and older), the population has almost doubled to about 200 million. For the long-term population to remain stable, natural mortality must eliminate about 100 million grackles between June and the following April for the population to begin the next annual cycle at 100 million birds (Dolbeer et al. 1997*b*, Dolbeer 1998; Fig. 7.1).

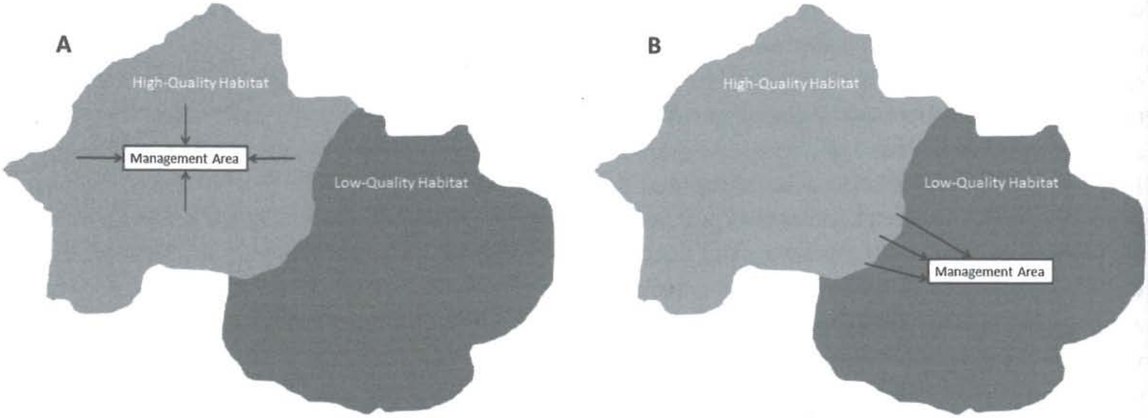

Fig. 7.2. Example of how surrounding habitat quality can affect management of overabundant populations. Management is conducted on a small scale in (A) a source population and (B) an adjacent sink population. Arrows indicate recruitment of new individuals into the management area.

In years when natural factors (e.g., inclement weather, disease) increase mortality or decrease reproduction, intraspecific competition may be reduced, with wildlife populations typically responding with increased survival or reproduction. Conversely, if natural factors result in an exceptional year of successful reproduction or low mortality, subsequent increased competition for food and habitat typically reduces reproduction or survival. These compensatory factors (Caughley 1977) dampen fluctuations in annual population levels and can stabilize the population in the long term. Exceptions occur with fundamental changes in habitat quality or mortality/reproductive factors. For example, the dramatic increase in the double-crested cormorant (*Phalacrocorax auritus*) population in the Great Lakes in the 1980s and 90s resulted from the combination of increased reproduction (elimination of chlorinated hydrocarbon pesticides) and decreased mortality (protection by Migratory Bird Treaty Act and enhanced food supply through the introduction of large-scale fish farming in the southern USA; Hatch 1995). Many other large bird species in North America and Europe exhibited similar increases in populations from 1980 through 1999 because of fundamental changes in carrying capacity (Dolbeer and Eschenfelder 2003).

Understanding the factors contributing to population fluctuations is especially relevant to the management of overabundant populations. Managing such populations in small areas may achieve temporary reductions, but these reductions may fail over the long term if the spatial and temporal scales and the factors governing dynamics at those scales are not considered. In a hypothetical simple source–sink system where high-quality habitat represents a source of individuals and low-quality habitat represents a sink, management of an overabundant population at a small scale will likely require repeated removals over multiple years because (1) removed individuals within the management unit in the source population will be replaced by recruitment from the surrounding population (Fig. 7.2A), and (2) individuals removed from the management unit embedded within the sink population will be replaced (possibly at a slower rate) by recruits from the adjacent source population (Fig. 7.2B). Under this scenario, one viable management option may be collaboration between airport managers and biologists with local municipalities and land owners to reduce desired habitat to less desired habitat for those species being managed (e.g., Blackwell et al. 2009*a*). Because the risk to aviation safety must be mitigated at airports, the removal of wildlife that disperse into the air operations area (AOA), even when habitat management and harassment programs are in place to discourage such dispersal, is often an ongoing part of the airport's WHMP (see case studies below).

Another aspect of population dynamics, one applicable to management of wildlife populations that pose hazards to aviation, involves the concept and practice of reproductive control to manage overabundant wildlife populations that are causing conflicts with humans.

Because the urbanized public generally advocates non-lethal means of managing problem populations of wildlife, there has been increased interest in the development of reproductive control strategies for wildlife species (Fagerstone et al. 2010). However, the modeling of population responses to various levels of lethal and reproductive control clearly demonstrate that for almost all species, lethal control is more efficient in reducing populations than reproductive control (Dolbeer et al. 1988, Dolbeer 1998, Blackwell et al. 2002). The exceptions are some small rodent and bird species with high reproductive rates and low survival rates (Dolbeer 1998)—species that pose little hazard to aviation (Dolbeer and Wright 2009). That reproductive control (e.g., oiling eggs in nests of gulls [*Larus* spp.] or Canada geese) may take several years to reduce the target population size makes this approach unacceptable for solving immediate risks posed by wildlife to aviation.

Population Management to Reduce Wildlife Strikes at U.S. Airports: Case Studies

There are numerous situations in which lethal control has been implemented to resolve human conflicts with wildlife at airports. In 2011, U.S. Department of Agriculture biologists used some level of lethal wildlife control at 314 civil and military airports in the USA as part of integrated management programs (Begier and Dolbeer 2012). Lethal control also has been used frequently in other (nonaviation) situations, such as agriculture, to reduce human–wildlife conflicts (Dolbeer 1986, Bedard et al. 1995, Dolbeer et al. 1997*b*). The following three case studies from airports demonstrate the utility of lethal control as part of integrated management programs.

Gulls at John F. Kennedy International Airport

Gull–aircraft collisions have long been a serious problem at John F. Kennedy International Airport (JFK), New York, New York, USA. Gulls, of which 60% were laughing gulls (*L. atricilla*), caused 86% of bird strikes from 1988 through 1990, averaging 261 strikes per year. Laughing gulls are present from May through September in association with a nesting colony at Jamaica

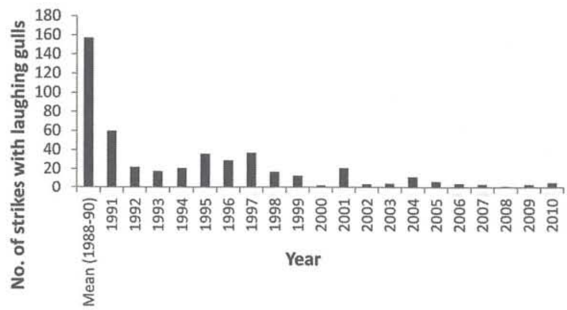

Fig. 7.3. Number of strike events involving laughing gulls at John F. Kennedy International Airport, New York, New York, USA (1988–2010). A shooting program was implemented in 1991.

Bay Wildlife Refuge, which is adjacent to the airport. Although the airport implemented numerous nonlethal actions to reduce gull presence at the airport in the 1980s, the number of strikes increased as the nesting gull population increased in the adjacent wildlife refuge (Dolbeer et al. 1993).

As an alternative approach to reduce strikes in 1991 (and continuing through 2011), biologists started a population management program in which managers stationed on JFK airport boundaries shot gulls flying over the airport from May through August. As a result of the shooting program, the number of strikes with laughing gulls was reduced to 38% of 1988–1990 levels in 1991 (the first year) and to 1–5% of 1988–1990 levels in 2008–2011 (Washburn et al. 2009, R. A. Dolbeer, unpublished data). Strikes by the three other gull species were reduced to 10–52% of preshooting levels over the same time periods. In 1991 and 1992, about 14,000 and 12,000 laughing gulls, respectively, were killed; this number declined to about 2,000–6,000 gulls in subsequent years (Washburn et al. 2009, R. A. Dolbeer, unpublished data). The laughing gull colony in Jamaica Bay has declined 73%, from 7,629 nests in 1990 to 2,040 nests in 2011 (Dolbeer et al. 1997*a*, Washburn and Tyson 2011). That the colony size declined by 73% from 1990 to 2011 while the annual strike rate of laughing gulls declined by over 95% (2008–2011; Fig. 7.3) indicated that many laughing gulls altered flight patterns and avoided the airport in response to shooting (Dolbeer et al. 2003). Although the shooting program has reduced the local population of gulls flying over JFK (Fig. 7.4), the regional population (>300,000 birds),

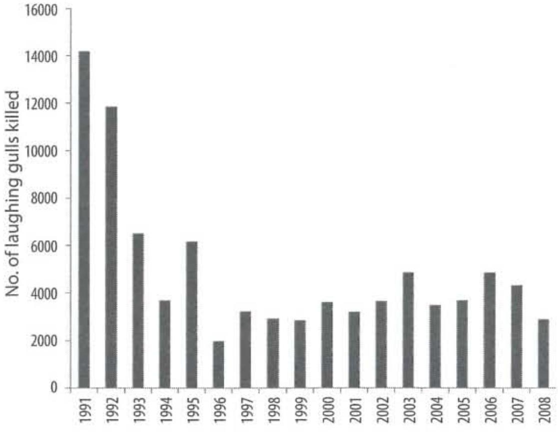

Fig. 7.4. Number of laughing gulls killed by shotgun at John F. Kennedy International Airport, New York, New York, USA (1991–2008).

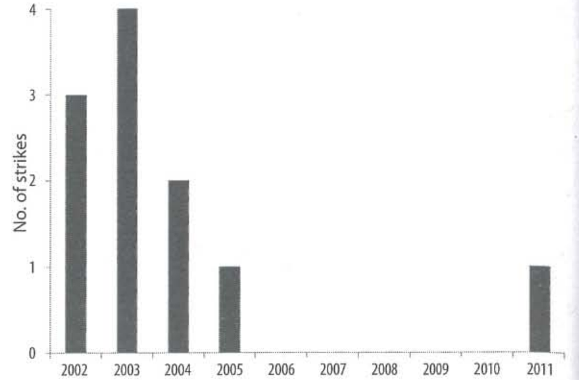

Fig. 7.5. Number of Canada goose strikes with aircraft <152 m (500 feet) above ground level at LaGuardia Airport, New York, New York, USA. Years comprise the twelve months from July through July (e.g., 2002 is July 2002 through June 2003); 2011 represents July–December only. From June 2004 through June 2011, 1,456 geese were removed from Rikers Island.

as predicted by modeling, has not been negatively impacted (Dolbeer 1998, Dolbeer et al. 2003). This study demonstrated that shooting can significantly reduce gull–aircraft collisions at an airport by both reducing the local population (but not the regional population) and altering flight patterns of surviving gulls.

Canada Geese near LaGuardia Airport

The resident (nonmigratory) population of Canada geese increased dramatically in North America from about 0.25 million in 1970 to 3.47 million in 2010 (Dolbeer and Seubert 2011), posing a substantial hazard to aircraft (Dolbeer and Eschenfelder 2003). In the 1990s, a portion of the growing population of resident Canada geese in New York City began using Rikers Island as a gathering site during the molting season (June–July). Rikers Island is located in the East River, about 0.5 km (0.3 miles) from LaGuardia Airport (LGA), New York, New York, USA. During the two-year period from July 2002 to June 2004, seven Canada goose strikes were recorded at LGA (all at <152 m (500 feet) above ground level; Fig. 7.5). These strikes included a passenger aircraft departing LGA in September 2003 that hit at least five Canada geese, causing an uncontained failure in one engine and requiring an emergency landing at JFK, 18 km (11 miles) away (National Transportation Safety Board 2004).

As a result of these strikes, a population management program was initiated at Rikers Island in June 2004 in which 518 resident geese, representing over 90% of the geese using the island, were rounded up during the molt (when they are flightless) and euthanized. In the seven subsequent years, the number of geese removed from the island steadily declined to 55 in 2011 (Fig. 7.6). The number of strikes at LGA involving Canada geese at <152 m above ground level (and thus in the airport environment) also declined in the aftermath of the management program (Fig. 7.5). Compared to the seven strikes recorded in the two years before the first removal at Rikers (June 2004), there have been only four strikes in the subsequent seven years. Two of those four strikes occurred in August–September 2004, less than three months after the first removal; there have been only two strikes in the subsequent seven years (October 2004 to December 2011). This focused population management program resulted in a major reduction in the local population of Canada geese near the airport and the number of strikes by this high-risk species. This program, involving the removal of 1,456 geese from 2004 to 2011, has had no impact on the regional population. The metropolitan area of New York City currently contains 15,000–20,000 resident Canada geese (B. Swift, New York State Department of Environmental Conservation, personal communication; Collins and Humberg 2011).

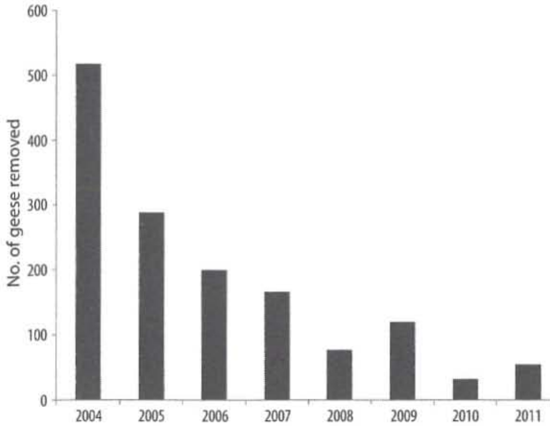

Fig. 7.6. Number of Canada geese removed from Rikers Island, New York, New York, USA, during the molt period (late June) from 2004 through 2011. Each year, >90% of the geese on the island were captured and euthanized.

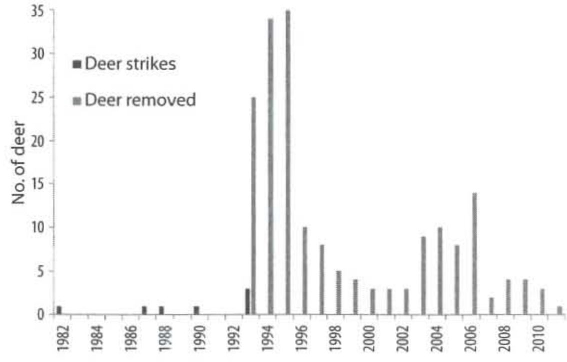

Fig. 7.7. Number of aircraft collisions with white-tailed deer at Chicago O'Hare International Airport, Chicago, Illinois, USA (1982–2011). In December 1993, 25 deer were shot and removed at night. From 1994 through 2011, 160 additional deer were killed. There has not been a deer strike at O'Hare since November 1993.

Deer at Chicago O'Hare International Airport

Deer at airports pose one of the highest risks of any wildlife species to departing and arriving aircraft (Wright et al. 1998, Dolbeer and Wright 2009, Biondi et al. 2011, DeVault et al. 2011, Dolbeer et al. 2012). Deer-proof fencing is the best long-term approach for excluding deer from AOAs; Chapter 5). However, larger airports may require >15 km (9 miles) of fencing to secure the AOA, often traversing uneven ground with numerous gates and culverts. Even with good fencing, it is not uncommon for deer to enter AOAs, especially in areas with high deer populations (DeVault et al. 2008). From 1990 through 2010, civil aircraft struck about 1,000 deer (*Odocoileus* spp.) at airports in the USA (Dolbeer et al. 2012).

In 1993, aircraft struck three deer at Chicago O'Hare International Airport (ORD), Chicago, Illinois, USA, prompting emergency action. In December 1993, sharpshooters removed 25 deer from the AOA at night, followed by the removal of 34, 35, 10, and 8 deer in 1994, 1995, 1996, and 1997, respectively. By 1998, the perimeter fence had been improved substantially to exclude deer, but deer still occasionally entered the AOA. Up to 14 deer were removed per year from 1998 to 2011 (Fig. 7.7). When appropriate, deer removed from the airport were processed and donated to charitable organizations.

The combination of lethal control starting in December 1993 and improved fencing resulted in no deer strikes at ORD in the subsequent 19 years from November 1993 through 2011. The overall deer population density in the Chicago area has not been estimated but is considered high (Etter et al. 2002); in 2005, Cook County (where ORD and Chicago are located) had about 1,000 deer–automobile collisions, the highest of any county in Illinois (Flood 2008). The overall deer population in Illinois is about 800,000, with over 150,000 harvested by hunters annually (Channick 2010); clearly, the removal program at ORD has not adversely affected local or regional deer abundance.

Summary

Lethal management of wildlife on and near airport properties is often an essential component of integrated management actions to mitigate the risk of wildlife–aircraft strikes. Despite the potentially catastrophic consequences of wildlife strikes, however, lethal management often evokes contention from the public. Management decisions involving population reduction must therefore be based on (1) an understanding of the factors affecting wildlife population dynamics, (2) the integration of lethal management with nonlethal methods, and (3) observational data before, during, and after implementation. These observational data (numbers killed, population levels, and number of

strikes with aircraft) are critical to determine the impact of lethal management actions on each wildlife species' population and on the mitigation of risk to aircraft using the airport. This information should be compiled into periodic reports (typically annually) that are made available to the public.

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