

**STATUS OF THE DOUBLE-CRESTED CORMORANT
(*PHALACROCORAX AURITUS*) IN NORTH AMERICA**

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EXECUTIVE SUMMARY

Introduction: Since the late-1970s, numbers of Double-crested Cormorants (*Phalacrocorax auritus*) (DCCO) have increased significantly in many regions of North America. A variety of problems, both real and perceived, have been associated with these increases, including impacts to aquaculture, sport and commercial fisheries, natural habitats, and other avian species. Concern is especially strong over impacts to sport and commercial fishes and aquaculture. Because of increasing public pressure on U.S. government agencies to reduce DCCO conflicts, the USFWS is preparing an Environmental Impact Statement (EIS), and in conjunction with the U.S. Department of Agriculture/Wildlife Services (USDA/WS) and state resource management agencies, will develop a national management plan for the DCCO. This assessment will be used to prepare the EIS and management plan.

Populations and trends: The DCCO breeding range in North America is divided into five geographic areas. Since at least 1980, numbers have clearly increased in three of the breeding areas: Canadian and U.S. interior, Northeast Atlantic Coast and Southern U.S. In these populations, much of the growth occurred between the late 1970s – early 1990s; from the early 1990s – 2000 growth rates have slowed or appeared to stabilize in many states and provinces. For the Pacific Coast and Alaskan breeding populations it was not possible to summarize trends overall because recent data for birds breeding in significant portions of these regions (e.g., Alaska, Mexico) are not available, or have not been collected in a coordinated and timely fashion for the populations as a whole. Along some parts of the Pacific Coast, breeding numbers declined in the 1990s (e.g., British Columbia, species is listed as Vulnerable and is being considered for Threatened status). In other areas significant increases occurred. Concurrently, numbers also increased on the wintering grounds, particularly in the Mississippi River Delta region, an area of high human-cormorant conflict over catfish resources.

Many historical records from across the continent indicate that the species was or may have been more abundant and widespread than is currently presumed. While most of these early accounts are largely qualitative, many report huge numbers of cormorants, suggesting that recent population increases may represent recovery towards historical (presettlement) levels in certain regions. In some areas where the DCCO has been documented as a recent breeder, the species is actually re-colonizing after an absence of 50 – 300 years.

Reasons for population increases: There appear to be five major factors that led to dramatic increases in DCCOs in North America since about 1970. These include:

1. Ban on DDT (1972) and other pesticide reduction regulation. Prior to this time (but post WWII) widespread use of DDT occurred. Cormorants accumulated high levels of DDT through their food supply, which interfered with reproduction. Depressed populations began to increase after DDT was banned.
2. In 1972 the DCCO was added to the Migratory Bird Treaty Act protected bird list. Before 1972, federal legislation did not prevent killing or harassment of cormorants during their annual cycle. Some states also provided special protection for DCCOs around this time.

3. Human induced changes (e.g. accidental and intentional introduction of exotics; over fishing; changes in water quality) in aquatic communities in the breeding range.
4. Development of aquaculture (e.g. catfish farms) in the south (especially Mississippi Delta region) that provided a new food source.
5. Creation of additional breeding and foraging habitat (e.g. reservoirs; dredge spoil islands).

Diet and native fish populations: DCCO diet is characterized by great temporal and spatial variation. The DCCO is known to feed on > 250 species of fresh and saltwater fishes. Cormorants are generalists and eat abundant fish in the size range 3 – 40 cm; < 15 cm is preferred. Review of diet studies (> 40) indicates most sport and commercially valuable fish species do not contribute substantially to DCCO diet. Though there are exceptions, most studies conclude that sport and commercially valuable fish species are not negatively impacted by DCCOs, and that DCCOs have minor effects on fish populations relative to human harvest and other mortality factors. The most common claim against DCCOs is that they reduce sport or commercial catches, but the actual relationship between cormorant predation, fish population size and human harvest is poorly understood. This lack of information contributes to the complexity of cormorant-fish-fishery interactions.

Rigorous quantification of cormorant predation on fish populations or on subsequent sport or commercial catches requires more precise estimates of several key parameters, including: prey fish population sizes; prey fish mortality sources and rates; age class distribution of fish consumed. Additionally, a better understanding of compensatory processes within prey fish populations is essential (e.g., predation may reduce competition so that remaining fish survive longer or younger fish grow faster). However, no study conducted so far has obtained robust estimates for all of these parameters. Therefore, while DCCOs may cause fish populations to decline, none of the studies reviewed provided data rigorous enough to demonstrate that they do so. The effect of cormorant predation can be either compensatory (if the cormorants do not eat them, the same proportion may be removed by other factors) or additive (mortality due to cormorant predation is not replaced by another factor). However, investigators have rarely examined cormorant predation in the context of other mortality or limiting factors.

Because of great spatial variation in DCCO diet and unique complexities of individual aquatic ecosystems, DCCO predation impacts need to be assessed locally. To do this biologists need a more comprehensive understanding of local fish population dynamics and standardized methods for assessing cormorant diet.

Diet and aquaculture facilities: Studies show DCCOs may eat large numbers of catfish locally and temporally. However, no study has quantified the economic impact on net harvest. Only one study has examined the issue of additive and compensatory mortality and concluded that mortality due to DCCO predation impacts were additive under certain circumstances, but insignificant in others.

Impacts on vegetation: Most colonial waterbirds destroy vegetation at breeding and / or roosting sites to some extent, and cormorants cause some of the most dramatic change. Cormorants impact vegetation through deposition of guano (excrement) that kills underlying vegetation and eventually trees, and through nest building behavior when they strip leaves and

small branches. In the short term these changes are of greatest concern if they affect rare plant communities or private property. From a long-term perspective these changes may be insignificant on an ecosystem scale. Few studies have been conducted to characterize and quantify vegetation change due to cormorant nesting and roosting habits.

Impacts on other bird species: DCCOs are hypothesized to have two potential effects on other colonial waterbird species: competition for nest sites and habitat degradation. Direct interspecific competition for nests and nest sites may occur but has not been documented through careful study. Most impacts appear to occur indirectly through habitat degradation (e.g. defoliation, tree die-off). While there is some evidence that DCCOs may displace other species, no studies have clearly established DCCO impact on other birds at even a colony level scale.

Management options: Humans have attempted to manage cormorant numbers in the western hemisphere for at least 400 years. Currently in the U.S. all lethal take requires permits from the USFWS, except at aquaculture facilities in those states under the 1998 Federal Depredation Order. Depredation permits can be obtained to prevent economic impacts or impacts to endangered, threatened or species of conservation concern. Non-lethal harassment of birds depredating or about to depredate does not require permits. To reduce cormorant impacts primarily to fisheries, aquaculture, vegetation and other colonial waterbirds, a large number of techniques has been developed or proposed. These techniques utilize lethal and non-lethal measures and may be used at local, regional or population levels. The effectiveness of these measures is difficult to assess because in many cases impacts have been poorly quantified.

Most techniques used at the local level are non-lethal. Lethal control may help reinforce local non-lethal control techniques. However, because cormorants are highly mobile, lethal control at the local level may be ineffective at decreasing local populations. Although economic effectiveness cannot be assessed for individual control techniques, some appear more effective than others; future research should focus on reducing the costs of the most promising techniques. Many techniques have been poorly investigated; therefore conclusions about their economic and numerical effectiveness may be premature. Because aquaculture ponds are high quality foraging sites (high fish densities; lack of escape cover), control of cormorants on the breeding grounds is unlikely to eliminate the need to practice local control. To make aquaculture ponds less desirable foraging sites, some form of control at the local level (e.g. exclosures, harassment) will likely still be needed. Previous efforts indicate that population control in general must be large scale and will require sizable human and capital inputs to be effective. Additionally, potential density dependent effects that compensate for control related mortality are poorly understood. Addressing these and numerous other uncertainties will enhance the development of a scientifically based, large-scale population control effort.

Finally, no control is a management option that is economically justified if the costs of control are greater than the losses associated with cormorant impacts.

Population Models: Models have identified data gaps critical for understanding population dynamics and predicting control effectiveness; modeling is potentially a very strong tool for gaining insights into cormorant management. Prediction of future DCCO population trends and analysis of control methods is hampered by lack of age-specific data for this species. More effort

needs to be put into obtaining data needed to strengthen model predictions, and increased effort should focus on predicting management outcomes and follow progress. Until better data are available, however, such modeling efforts should include rigorous sensitivity analyses to investigate uncertainties in parameters used and assumptions made in the model.

Current research and monitoring efforts: Of 33 U.S. states and nine Canadian provinces to which surveys were sent, nine reported research in progress and 19 have monitoring programs. Research addresses: cormorant diet, bioenergetics, impacts to aquaculture, sport and commercial fisheries, foraging range and foraging behavior. Additional studies are attempting to determine effectiveness of harassment at day and night roosts, effectiveness of barriers at aquaculture ponds, and nutrient enrichment in aquatic and terrestrial habitats. A satellite telemetry study will determine migration patterns, breeding locations and winter movements of cormorants at catfish farms. All monitoring efforts are used to determine population distribution and trends.

Future research priorities: The assessment identified many research needs. Highest priority studies on DCCOs fall within the following broad topics: (1) demography, (2) impacts on fisheries and aquaculture, (3) management techniques, (4) impacts on flora and fauna and (5) distribution.

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INTRODUCTION

Background Information

Since the 1970s, numbers of Double-crested Cormorants (*Phalacrocorax auritus*) (DCCO) have increased significantly in many regions of North America. A variety of problems, both real and perceived, have been associated with these increases, including impacts to aquaculture, sport and commercial fisheries, natural habitats, and other avian species. Concern is greatest over impacts to sport and commercial fishes, and to aquaculture. Impacts to the latter appear more significant, but studies thus far have not determined the magnitude of these impacts.

The DCCO is protected in the United States by the Migratory Bird Treaty Act (MBTA) of 1918, as amended (16 U.S.C. 703-712). DCCOs were added to the list of protected species (50 Stat. 1311, T.S. No. 12) in 1972 following amendment of the U.S. Convention with Mexico (23 U.S.T. 260, T.I.A.S. 7302). DCCOs are not included in the U.S. Convention with Great Britain, acting on behalf of Canada for the Protection of Migratory Birds (39 Stat. 1702 T.S. No. 628), and are therefore not protected by the Canadian Federal Government, although they receive protection by provincial governments. In addition to U.S. federal protection, DCCOs may also receive some protection at the state level.

Due to various concerns, legal actions have been taken to control DCCO numbers in the U.S. In March, 1998, USFWS established a depredation order (Trapp 1998) in response to complaints from catfish farmers and baitfish dealers. This order allows those engaged in commercial aquaculture to shoot cormorants without a federal permit at freshwater aquaculture premises or state-operated hatcheries in Minnesota and 12 southeastern states. Cormorant control programs also exist in individual states to control numbers to reduce their impacts to island vegetation and other colonial waterbirds, but these can only be carried out under the terms of a federal permit.

Illegal actions to control cormorants have also occurred. In the summer of 1998, frustrated fishermen who believed cormorants were responsible for game fish declines in the eastern basin of Lake Ontario illegally shot an estimated 1500 – 2000 Double-crested Cormorants on Little Galloo Island, Lake Ontario. In 1999, the State of New York requested a depredation permit to reduce cormorants on eastern Lake Ontario primarily for the purpose of controlling their predation on smallmouth bass, a popular sport fish. The State's secondary concern was to limit cormorant competition with other bird species. The USFWS issued a permit to prevent all reproduction through egg oiling on Little Galloo Island, and to continue nest destruction efforts on other islands on Lake Ontario to benefit other bird species and their habitats. In Canada, legal and illegal control for similar reasons has also occurred.

A resolution was signed by George W. Bush, then the governor of Texas, asking USFWS to evaluate the economic impact of the DCCO on sport fishing and to "consider removal of the Double-crested Cormorant from the protection of the Migratory Bird Treaty Act if national economic losses warrant severe control methods to keep the cormorant population at a manageable level within a given region." Because of these strong concerns over potential impacts associated with increases in DCCOs, the USFWS is preparing an Environmental Impact Statement (EIS), and in conjunction with the U.S. Department of Agriculture / Wildlife Services (USDA/WS) and state resource management agencies, plan to develop a national management plan for the DCCO. This status assessment will serve as a primary resource to aid in the EIS and development of the management plan.

The Status Assessment

To assess the status of the Double-crested Cormorant, we reviewed natural history, population data and trends, diet studies, cormorant impacts to vegetation and colonial waterbirds, cormorant control, population modeling, current research needs and future research priorities. Much of this information was obtained through a review of published literature, literature in press, and unpublished reports. In addition to summarizing information, we evaluated some studies (mainly those which attempted to determine impacts of DCCO predation) for accuracy and thoroughness. Similar increases in numbers of the Great Cormorant (*Phalacrocorax carbo*) (GRCO) have occurred in Europe, and similar conflicts with human interests have arisen. Because the DCCO and the European GRCO are ecological counterparts, and much significant research has been conducted on the latter species, we incorporated relevant insights gained through study of the GRCO.

We also conducted two surveys to obtain information. We first conducted a DCCO survey to obtain data on breeding, wintering and migration numbers and distributions, population trends, and current research, monitoring and management / research priorities; and to obtain information on the perceptions of impacts to natural resources and damage problems associated with DCCOs in each state / province. We sent this survey to agency (both government and non-government) wildlife biologists, university and museum biologists, and to USDA/WS personnel in areas where problems associated with cormorants are high. Survey recipients were asked to identify a fisheries biologist to whom we could address specific questions regarding impacts to sport and commercial fish. We then conducted a second survey with fisheries biologists to obtain more information on perceptions of impacts to fish and to gain a wider perspective on this problem. Where questions arose regarding information provided in the surveys, we made follow-up calls and tried to resolve unclear issues.

PART I: NATURAL HISTORY OF THE DOUBLE-CRESTED CORMORANT, *PHALACROCORAX AURITUS*

[This section is a brief summary based largely on the *Birds of North America* species account of the Double-crested Cormorant (Hatch and Weseloh 1999). Parts of some sections (breeding and winter range) come directly from this account; for a full treatment of the natural history of this species and for specific data sources the reader should see account. Other literature and information sources were also occasionally included.]

GENERAL INFORMATION

Taxonomy

The Double-crested Cormorant (DCCO) is in the order Pelecaniformes, family Phalacrocoracidae. Worldwide, between 30 – 40 species of cormorants are recognized. In earlier taxonomies most species were placed in the single genus *Phalacrocorax*; recent accounts favor dividing the family into two main groups, the typical cormorants and the strictly maritime ones, the “shags.” In North America, there are six species of cormorants, including the Double-crested; these include Great Cormorant (*P. carbo*), Neotropic Cormorant (*P. brasilianus*) [formerly called Olivaceous Cormorant (*P. olivaceus*)], Brandt’s Cormorant (*P. penicillatus*), Pelagic Cormorant (*P. pelagicus*), and Red-faced Cormorant (*P. urile*). The range of the Double-crested overlaps those of Brandt’s and Pelagic Cormorants on the Pacific Coast, from southern Alaska to the Baja Peninsula; the Red-faced Cormorant mostly along the coast of southern Alaska; the Neotropic in Texas, Louisiana, and w. Mexico, mainly Sinaloa; and the Great Cormorant along the east coast, from Newfoundland to Florida. The Double-crested is most closely related to the Neotropic Cormorant and is in the same subfamily (genus) as the Great (*carbo*) and Brandt’s (*penicillatus*) cormorants. The other subfamily includes species provisionally named as shags.

Five subspecies of the Double-crested Cormorant have been described, based on size and crest characters: 1) *P.a. auritus* (Northern Double-crested Cormorant), most numerous, breeds widely in the interior and on the northeast coast; 2) *P.a. cincinatus* (formerly White-crested or White-tufted Cormorant) occurs in Alaska; 3) *P.a. albociliatus* (formerly Farallon Cormorant) breeds on Pacific Coast and inland, possibly to New Mexico, Utah and Montana; 4) *P.a. floridanus* (formerly Florida Cormorant) is resident in Florida and Caribbean; 5) *P.a. heuretus* (no common name) is resident on San Salvador I., and possibly other islands of the Bahamas (see figures in Palmer 1962, p. 331; Johnsgard 1993, Fig. 51, p. 200). Characteristics of birds breeding in Mexico have not been established. Banding recoveries suggest that little mixing occurs across the Rocky Mountains.

Physical Description

The DCCO is a medium to large (typical length 70 – 90 cm, body mass 1.2 – 2.5 kg) dark waterbird. Cormorants have elongated bodies, moderately long necks and bills, long wedge-shaped tails, and totipalmate feet. Resting birds often hold wings in a spread-wing posture, thought to aid in drying wet feathers. Distinctive characteristics of the DCCO include brilliant turquoise eyes, a golden throat pouch, elegantly patterned back feathers, and for a short time prior to breeding season, a cobalt-blue mouth. Plumage is black or dark brown with a dull

greenish or bronze gloss. There is considerable variation in body size, and in color and size of crest. Colors of eye-lid may vary geographically or individually from blue to orange. The genus name, *Phalacrocorax*, is Greek for “bald-headed raven”; the species name, *auritus*, is Latin for “eared” and refers to the crests above the eyes, feathers which are fully developed only for a short time early in the breeding season.

Range

Widely distributed in North America, but not elsewhere. Vagrants have been reported from England 1989, and the Azores in 1991.

Breeding Range.

Five major breeding zones have been defined:

- 1) Alaska
- 2) Pacific Coast
- 3) Canadian and U.S. Interior
- 4) Atlantic Coast
- 5) Florida and the western Caribbean

The five breeding zones are partly reflected in subspecific designations. Populations of *P.a. auritus* have expanded and contracted from two areas: freshwater breeders in the Prairie provinces, and maritime birds in the northwest Atlantic. The other four subspecies are concentrated in Alaska (*P.a. cincinatus*); along the Pacific Coast (*P.a. albociliatus*); in Florida and the Caribbean (*P.a. floridanus*); and in the Bahamas and Cuba (*P.a. heuretus*). The Alaskan population breeds at Nunivak Island., and se. Bering Sea from e. Aleutian Islands. to the southeast coast, including Kodiak Island. The Pacific Coast population breeds between s. British Columbia and Sinaloa, Mexico; most breeding is coastal, though some occurs inland. The Canadian and U.S. interior population breeds from n. Alberta through central Ontario; James Bay and sw. Québec, south to central Utah, central Colorado, w. central Nebraska, se. South Dakota, w. central Minnesota, ne. Iowa, central Wisconsin, and n. Lower Peninsula of Michigan; range extends west to sw. Idaho and east along GL to lower St. Lawrence River. Also breeds locally in other areas within the interior (see Hatch and Weseloh 1999). On the Atlantic Coast, breeds from Newfoundland to New York, and small numbers elsewhere (see Hatch and Weseloh 1999). Most breeding is coastal. In the southeast, resident in Florida, and occurring locally in small numbers along Gulf Coast to Texas. Scattered residents in Cuba, and uncommon resident in Bahamas. Also nests on coast of Yucatan Peninsula and n. Belize. Recent expansion has led to blurred boundaries for Zones 3, 4 and 5. For further discussion and delineation of the five breeding zones, see Part II, Biologically Relevant Units and Organization of Data.

Winter Range.

On the Pacific Coast and Alaska, chiefly resident, though some dispersal occurs. Birds breeding in the interior and on the Atlantic Coast are strongly migratory, and most winter coastally from North Carolina to the western Gulf of Mexico. Also winters along the Gulf of Mexico from Tamaulipas south to the Yucatan Peninsula and Belize. Inland, significant numbers occur at lakes, rivers and impoundments; regularly winters inland from Atlantic Coast along major rivers to se. Pennsylvania, central Maryland, and throughout lower coastal plains of

Virginia and the Carolinas. Also winters inland from Gulf Coast (especially along the Mississippi and other rivers) north to central Georgia, n. Tennessee, sw. Indiana, s. Illinois, n. Arkansas, e. Kansas, eastern half of Oklahoma, Texas (except for Panhandle). Small numbers are increasingly observed in coastal New England, the Maritimes, the Great Lakes, and the prairies. Other populations breeding in Florida, Mexico, Cuba, and the Bahamas are resident.

Habitat

Breeding Season Habitat Requirements.

Colonies require sites safe from ground predators and close to feeding areas (usually < 10 km). Ponds, lakes, slow-moving rivers, lagoons, estuaries and open coastlines are utilized. Where available, selects small rocky or sandy islands. May also use artificial sites such as bridges, wrecks, abandoned docks or purpose-built towers. Though nests on ground or in trees, ground nesting may be the ancestral and preferred habit; tree nesting may be response to predators. Trees used for nesting are usually standing in or near water, on islands, in swamps, or at tree-lined lakes. Where predators are present, depends on flooded snags or live riparian trees. Also nests on emergent vegetation in marshes. In all seasons requires suitable places for nighttime roosts and daytime resting or loafing. Roosts and resting places are often on exposed sites such as rocks or sandbars, pilings, wrecks, high-tension wires or trees near favored fishing sites.

Winter Habitat Requirements.

Largest numbers of wintering birds occur along the southern coasts, and require similar characteristics in feeding, loafing and roosting sites as when breeding. Cormorants wintering along the North Carolina coast roost in or near the inlets on sandbars in dense flocks of up to 5,000 – 10,000 birds, or more, so that the sandbars are often blackened. They roost so close they are nearly touching each other. Sandbars or shoals chosen are high enough that they are not underwater at high tides. During the day, birds feed in large flocks, often numbering several thousands, in the inlet, sounds, and inshore ocean. Very little roosting is in trees, but some roost on channel markers, pilings, etc. Birds also rest on sandbars and markers during the day. They are often seen flying to sandbars, and then flying out to sea in the afternoon; at dusk they return to the sandbars (H. LeGrand, pers. comm.). Around aquaculture sites winter numbers are increasing, and in Mississippi many that winter near catfish farms roost in isolated cypress swamps. There may be differential selection between sexes in winter habitat use; in Mississippi, males predominated in samples shot at inland roosts (Glahn et al. 1995), and in Texas females predominated in a sample collected in Houston Ship Channel (King et al. 1987).

BIOLOGY

Migration

Atlantic Coast and interior nesting birds are highly migratory; all age groups migrate. In other areas, the species is mostly resident within the breeding range and migratory habits are poorly understood. More northerly populations breeding in Alaska may migrate south to s. British Columbia and Washington.

Migrants wintering in the Gulf of Mexico begin heading northward as early as February. Along the eastern seaboard they follow the Atlantic coastline; in the interior they follow river systems and fly overland. Earliest individuals reach Oklahoma and Virginia by 4 – 5 March;

Massachusetts, the southern Great Lakes, Minnesota, S. Dakota, and s. Idaho by late Mar – early April; upper Great Lakes by early to mid April; more northerly destinations, e.g., the prairie provinces, coastal Maine, are not reached until mid-late April. In British Columbia, migration occurs mid-Mar to late May, peaking late April – early May. Analysis of banding recoveries indicates that first-year birds return north about a month later than older birds. Recently, some birds have been remaining year-round and breeding along the Gulf Coast.

Autumn migration is essentially the reverse of spring. Earliest autumn migrants appear to reach wintering grounds faster than they reached breeding grounds in spring. Migration underway on the Canadian prairies by mid-late August, at which time first migrants are arriving on the Upper Mississippi River, Ohio, Massachusetts, Virginia and Texas. By September, migration well underway throughout range. Massachusetts, Rhode Island and Minnesota report peak numbers of transients in October. Most banded migrants recovered in Oct and Nov. Along coastal New England, bulk of migration observed 25 September – 17 October. Individuals arrive on winter grounds September to November. Some fly across open ocean.

Inland, migrates in small flocks, < 50 – 100; along coasts, near shore flocks may consist of 1000s of birds. Migratory flight begins soon after dawn and continues all day, with flocks observed flying late in evening.

Reproduction

Usually forms dense breeding colonies (100s – 1000s); solitary nests very rare. Often nests in mixed-species colonies. DCCOs are believed to have extreme fidelity to colony sites. Many young first breed at natal colonies, and philopatry to proximity of natal colony is probable; in banded birds recovered in June that were at least 3 years old, the median distance to natal colony was only 25 km.

In British Columbia, most DCCOs began breeding as 3 year olds (in their 4th summer). However, a small portion of color banded birds first bred as 1 year olds (4.7 %) and 2 year olds (16.5 %). Immatures (1 and 2 yr olds) are present on breeding colonies by June.

Ground nesting cormorants typically nest on low-lying rock islands or reefs, away from or with sparse vegetation. Arboreal nesters use evergreen and deciduous trees, initially alive but killed within 3 – 10 years due to guano deposition over time. Nesting substrates include limestone and granite bedrock, large boulders or fallen trees, gravel, cobbles, beach ridges, soil or standing trees. Possibly usurps attended or unoccupied nests of some herons and egrets. Nest characteristically includes finger-sized sticks and other bulky items collected from diverse locations, some several kilometers away. Along seacoasts seaweed forms the usual nest-base; flotsam and jetsam often incorporated, as are parts of dead birds. Lining is usually of grass, rootlets and similar materials. As breeding season progresses, the nest receives pebbles and bones from pellets, and a coating of guano on the outside that seals the nest together. Many nests are used repeatedly and built upon each year, becoming tall turret-like structures; some reach heights of 2 m or higher. Tree nests are usually much smaller but have a deeper inner cavity than ground nests. Ground nesters defend small area around nest within beak range (0.38 m). Reported inter-nest distances vary widely, from 3.3 – 220 cm.

Double-crested cormorants lay between 1 – 7 eggs. Average clutch size ranges from 2.7 – 4.1; modal clutch size is four. First eggs are laid 2 – 4 weeks after arrival to breeding colonies. In Ontario, egg laying occurs late Apr to Aug; in the St. Lawrence River estuary from about 10 May to Jul. Eggs are cylindrical ovate in shape. Shells are pale blue and unmarked, but pigmented layer often obscured by variable outer calcite cover that is initially white and porous.

Calcite layer has chalky texture and gives irregular surface. Egg mass is small compared to other seabirds; reported averages range from 44.9 – 46.5 g, 2.7 % of adult mass. Eggs are laid daily or on alternate days. Following loss of complete clutch, relaying is frequent, occurring within 19 days. However, successful second broods are rare. Eggs are extremely cold tolerant, but vulnerable to heat stress.

Incubation begins gradually, reaching a maximum intensity when third egg is laid. Eggs are incubated by lying on top of warm webs of feet; abdomen and breast are lowered onto them. Duration of incubation varies from 25 – 28 days. Both sexes incubate, but the female does more during the first half of the incubation period. Changeovers occur at intervals of 1 – 3 hours. Hatching occurs asynchronously, but intervals between eggs typically less than 1 day.

Chicks are altricial, and barely able to move at hatching. Eyes open at 3 – 4 days; egg tooth drops off between 4 – 7 days. Down appears within about one week. Thermoregulatory ability not complete until 2 weeks. Young are cared for equally by both parents. Brooding is nearly continuous for the first 12 days, but ceases with the appearance of down and effective endothermy. Young will remain in ground nests for about 28 days if undisturbed (though have walked from ground nests as early as 21 days when approached by humans or predators). Young in tree nests or in nests on cliffs may remain there until they are able to fly, which occurs between 6 – 8 weeks of age. Diving ability develops and plumage is complete at about the same time. Young are completely independent of parents at 10 weeks of age.

Annual reproductive success varies within and among colonies, but hatching success usually 50 – 75 %; fledging success 1.2 – 2.4 young / nest or 74 – 95 %. Chick loss from hatching to fledging is often low, e.g. 5 % in coastal B.C. All figures much lower for DDE-contaminated populations. In St. Lawrence estuary, reproductive success parameters lower for late nesting cormorants (June) compared to early nesters (May). Average lifetime production (lifetime reproductive success) calculated for birds breeding on Mandarte I., B.C. was 3.28 young (van der Veen 1973).

Foraging Ecology

The DCCO is an opportunistic, generalist feeder, preying mainly upon abundant, easy-to-catch fish species. Usually slow-moving or schooling fish, ranging in size from 3 – 40 cm but commonly < 15 cm, are taken. The DCCO appears to be strictly diurnal in its feeding habits, and usually forages in shallow water (< 8 m) within 5 km of the shore. The prey of Atlantic birds suggests that they are more likely to feed at the bottom of the water column, while that of Pacific and inland birds suggests that they feed in mid-water. Cormorants respond rapidly to high concentrations of fish and often congregate where fish are easily caught, such as put and take lakes, stocking release sites, aquaculture ponds, dams, and other areas where fish are concentrated.

To capture food, cormorants dive from the surface and pursue prey underwater. Prey is grasped in the bill and may be swallowed underwater. If the prey is large or difficult to handle, such as eels, flounders or spiny fish, it may be swallowed at the surface after first being shaken and hammered on the water. Prey is sometimes thrown in the air, caught and swallowed head-first. Frequently forages individually, but readily gathers to form feeding flocks (tens to hundreds). When feeding on schooling prey, sometimes loosely-coordinated foraging flocks are formed. This behavior has been observed most often in the breeding season and in late summer / early fall. Bottom-feeding is usually solitary.

Diet

Information on the diverse diet of this species has been recorded from as early as 1835, when Audubon (1835) observed that the food of Double-crested Cormorants consisted of “shrimps, lents, capelings, codlings, and other fishes”. In general, primary prey are forage species of little or no commercial value, but cormorants will take advantage of abundant species in the right size range. Over 250 fish species from more than 60 families have been reported as prey items. Occasionally, other aquatic animals, such as insects, crustaceans and amphibians are also taken. Diet is discussed specifically by region in PART III: DIET.

Longevity, Survival and Mortality

Oldest banded bird reported was 17 years, 9 months of age; however, wear and loss of aluminum bands is likely to lead to underestimates of survival based on recoveries. For birds banded as fledglings on Mandarte I., B.C., first year survival was estimated at 0.48, second-year at 0.74, and subsequent annual survival of 0.85; mean adult life expectancy 6.1 year (van der Veen 1973). Mortality factors for young birds include disturbance at breeding colonies, which can result in large mortality of hatchlings from exposure, and of eggs and young by predation; Newcastle disease can also cause significant mortality among young birds. Adult and large chick mortality factors include predation by Bald Eagles (*Haliaeetus leucocephalus*), entanglement in fishing gear, and shooting. Rates of mortality due to predation were not available. Fishing gear is reported as a major cause of death. Of birds banded in the Great Lakes between 1928 – 1995, 9 % of those recovered were shot; of band recoveries in Texas, 17 % were shot.

PART II. POPULATION DATA AND TRENDS

INTRODUCTION

Most population and trend data were obtained through the survey sent to wildlife biologists. We also utilized published literature, breeding bird atlases, unpublished reports and / or papers in press regarding trends and / or the history of cormorants in specific states and provinces. Survey recipients were asked to provide data on breeding, wintering, and migration concentrations. For breeding birds we asked recipients if cormorants were recent (1972 or later) breeders in their state / province, and to provide number of pairs, ownership of site (if colony \geq 300 pairs), and any productivity estimates. For wintering populations, we asked for number of birds and site ownership information (if concentrations $>$ 1000 birds). For migration concentrations, we asked for peak migration times and numbers and land ownership for significant stopover sites (concentrations of $>$ 500 birds). We also telephoned and e-mailed many individuals who provided us with data to get more precise information regarding numbers and trends in their states and provinces. Individuals completing surveys sometimes provided possible reasons for population increases; these were incorporated into the state and provincial profiles.

Breeding Birds

Biologically Relevant Units and Organization of Data

Five main zones for breeding Double-crested Cormorants in North America were defined by Hatch and Weseloh (1999) and these largely correspond to subspecies ranges (see Part I: Breeding Range.). In an effort to present biologically relevant information on biological units, or “populations”, we organized and presented population data based largely on these population zones, with a few modifications. In North America, populations of breeding cormorants occupy five main areas: Alaska (Zone 1); the Pacific Coast, from British Columbia to Mexico (Zone 2); the Canadian and U.S. interior (Zone 3); the Southeastern U.S (Zone 4), and the Atlantic Coast, from Newfoundland to New York (Zone 5) (Figure 1). For the most part, these zones are geographically or biologically (in terms of subspecies) distinct. Data were organized in this manner to aid and encourage individuals to view the biological unit, rather than political or organizational boundaries, when developing management plans.

These zones are thought to reflect fairly distinct breeding populations, but recent DCCO expansion and re-colonization has blurred the boundaries between these zones (Hatch and Weseloh 1999). Border states between interior and southeastern populations, interior and western populations, and southeastern and northeast Atlantic populations (e.g., Texas, New Mexico, Idaho, inland southern states, the Carolinas), qualify as “gray areas.” In these areas, it is not possible to determine precisely which zone breeding cormorants belong to; therefore, it was not possible to delineate absolute boundaries for each zone. However, because we were interested in presenting as much biologically relevant information as possible, and birds breeding in “gray areas” were in relatively small numbers, we determined the most likely population zone for each colony documented between 1970-2000 (Figure 1). These determinations were based on the best information available on subspecies range (Palmer 1962; Johnsgard 1993; Hatch and Weseloh 1999), and consideration of logical geographic units.

Additionally, to help with decisions for birds breeding in inland southern states (AR, OK, TN), southeastern coastal states (NC, SC), and southwestern states (TX, NM), we also

considered winter distribution, and the fact that birds breeding in many of the southern states are likely year-round residents, while birds from neighboring Zones 3 and 5 are migratory. The winter range in North America is quite distinct and occupies two main areas: in the eastern and central portion of the continent, the species winters mostly from Texas to Florida along the Gulf Coast, and along the Atlantic Coast through the Carolinas and Virginia; in western North America, the species winters mostly along the Pacific Coast, from southern British Columbia to Baja California Sur and the Gulf of California. Large numbers also winter inland in Texas, Oklahoma, and Arkansas. The large number of birds wintering in the eastern and central portion is a fairly easily defined, succinct, and very important unit in terms of human-cormorant conflicts. Therefore, it made the most sense to keep the southern states together and acknowledge the possibility that the small number of birds that breed in the boundary states of this zone may in fact be part of the interior or northeast Atlantic populations. For birds breeding in Idaho, where some have been found to possess characters intermediate between subspecies *P.a. albociliatus* and *P.a. auritus* (Burleigh 1972), suggesting that mixing occurs at the “boundaries” of Zones 3 and 5, we decided to rely on Burleigh’s (1972) conclusion that Idaho birds examined came closer to *P.a. auritus* than to *P.a. albociliatus*; thus we grouped Idaho birds with the interior population.

Therefore, when viewing these population zones, we suggest that special attention be paid to boundaries, especially “gray areas”. Extent and areas of overlap need to be more closely defined, especially if management actions are considered at a regional level or potential impacts to a population are under review. More detail is presented on population organization in the individual summaries for the zones; the problem is also addressed in PART VII. RESEARCH NEEDS.

Colony Locations

Survey recipients were asked to provide latitude-longitude coordinates for each colony location known since 1970. If these data were not available, we made follow-up calls and tried to determine approximate locations. We were able to obtain relatively good location data for most colonies and have included maps showing the distribution of colonies in each population zone during the last 30 years (Figure 1). In Zone 1 we show distribution of all known colony sites. In Zone 2 we show distribution of all known colony sites with the exception of interior California. D Shuford at Point Reyes Bird Observatory will supply a map for this area (D. Shuford, pers. comm.). In Zone 3, we were unable to obtain complete data for the following states and provinces: for Iowa, we were unable to obtain coordinates for 4 of 10 known colonies; for Minnesota, we were unable to obtain coordinates for 7 of 75 known colonies; for Alberta, Montana, North Dakota and Colorado we were unable to obtain any precise information. In Zone 4, we obtained coordinates for all known breeding colonies, with the exception of Florida, where coordinates were provided for “most” cormorant colonies in the state. In Zone 5, we obtained coordinates for all known colonies except for 5 of 12 on Prince Edward Island, and 2 of 9 in Newfoundland.

Maps showing distribution of active colonies by size at time of last survey are included for states and provinces for which we have the appropriate data in the five breeding zones. In Zone 4, the only state for which we produced this type of map was South Carolina. With the exception of Florida, for which the necessary data were not available, most states in this zone have marginal breeding numbers with few and relatively small colonies.

Trends

To determine trends in breeding numbers, we asked survey recipients to supply data on all known breeding colonies over 4 time periods: 1970, 1980, 1990, and 1998. Not surprisingly, many states and provinces did not have data for these specific years, so we asked for data from censuses conducted around these years. If that was not possible, we asked for whatever data were available. If complete counts were made for at least two years we generated population trends (Table 1, Table 4, Table 5, Table 6; see also figures in state and province summaries). To estimate percent average annual rate of change in number of pairs, we calculated $e^r - 1$, where

$$r = \frac{\ln(\text{recent count}) - \ln(\text{earlier count})}{\text{years between counts}} \quad \text{and} \quad e^r = \lambda$$

(Smith 1992). For figures we used all complete data available. For eastern and central North America (Zones 3, 4, 5), when no complete counts were available, we examined colonies within a state or province that had ≥ 2 years of data and compared the number of colonies that had increased at last count with the number that had decreased.

We also provided estimates of size of regional breeding populations for Zones 3 – 5 (Table 4, Table 5, Table 6), based on latest totals of breeding pairs in each state and province that made up the regional population. We caution that estimates for each state and province may utilize different survey methodologies, and may not all be obtained in the same year, thus regional estimates should be considered rough approximations at best. For Zones 1 and 2, comprehensive data were not available for an estimate of the entire region.

Because data were incomplete for many areas and / or time periods we did not attempt to calculate a rate of change for the continental or even regional populations. However, Tyson et al. (1999) reported that the number of DCCOs nesting in the U.S. and Canada increased about 2.6 % annually from 1990 through 1994. This trend was based on data that ranged from recent complete counts to conjectures based on old or incomplete information. While the mean percent annual change in the number of nesting pairs in those states and provinces that had recent complete counts was 16.2 %, the majority of states and provinces did not have recent complete counts, and thus Tyson et al. (1999) suggest that these rates of change be used with caution.

Wintering Birds***Locations and Winter Distribution***

Survey recipients were asked to provide latitude-longitude coordinates for all known wintering locations, including feeding and roosting sites where major concentrations (> 1000 birds) occurred. However, the majority of U.S. and Mexican states, and Canadian provinces do not conduct surveys for winter birds, and could not supply latitude-longitude information for DCCO wintering areas. Thus it was not possible to create a precise winter distribution map. We suggest that Hatch and Weseloh (1999) be consulted for winter distribution across the continent. To provide general information on wintering cormorants, we utilized data obtained through annual Christmas Bird Counts. A few states (AL, AR, and MS) provided locations for winter roosts (see below).

Use of Christmas Bird Count Data

The most comprehensive data for wintering DCCOs in the U.S. and Canada were obtained from annual Christmas Bird Counts (CBCs). Annual CBCs have been the only method

to consistently monitor winter birds on a broad scale. While CBCs are not undertaken to produce precise population estimates, they provide a useful index of population trends and valuable information on winter occurrence. For some states, CBC data provided the only information available about wintering cormorants. Therefore, for states with major wintering numbers (≥ 100 birds overwinter) we obtained CBC data (locations and numbers) collected between 1984 – 1997 for Zone 4 and 1984 – 1998 for Zones 1, 2 and 5. We also obtained data collected between 1984-1998 for states in Zone 3 that border Zones 2,4 and 5. CBC data were accessible for 1984-1998 at the time of our assessment (Sauer et al. 1996). For each state and province, we included all CBC sites where ≥ 100 birds were counted at least once during the period under consideration (see figures in state summaries). These sites are shown in Figure 2.

Wintering Numbers and Trends

In general, data on numbers of over-wintering birds were limited and it was not possible to determine trends. The information we provide on wintering trends is based mainly on “qualitative” assessment of apparent changes in numbers counted during CBCs. For some states (AL, BC, CA, FL, GA, KS, LA, ME, MA, MI, NJ, NY, NC, OK, OR, TN, TX, VA, WA) Sauer et al. (1996) analyzed CBC data (1959 – 1988) for trends, included in this assessment. CBC data (1989 – 1999) have not been analyzed, but are included to provide a broad picture of recent changes in winter numbers. CBC data should be interpreted cautiously, for a number of reasons (e.g., number of counts is not always consistent within states and provinces; observer skill level varies).

Limited survey data from Mississippi were included to provide a broader picture of recent changes in wintering numbers in this state, mainly in the Mississippi Delta region. Additionally, limited survey data from Alabama, Arkansas and Mississippi data were incorporated to compare CBC site distribution with known night roosts located by Wildlife Services surveys (Figure 51). Additionally, comments on perceived trends from biologists and other individuals or sources providing us with information on winter numbers were incorporated. At this time, it is not possible to provide an estimate for winter population size because data are incomplete. However, this may be an important estimate for management actions geared toward reducing impacts on the breeding and wintering grounds (see Non-breeding Birds and Total Population Size, below, and PART VII. RESEARCH NEEDS).

Migrant Birds

Migration / Stop-over Sites

Survey recipients were asked to provide latitude-longitude coordinates for all major (≥ 500 birds) migration stop-over sites. For most states, migrant numbers were very difficult to obtain because often no information was available, or migrants could not be distinguished from wintering birds. Some states provided general information about where concentrations of migrant birds were regularly seen, but because data were so limited we did not map them.

Trends

We provide limited information for a few states and provinces on trends in numbers of migrants in the specific state/provincial accounts.

Non-breeding Birds and Total Population Size

We did not obtain information on non-breeders, which include sub-adult and adult non-breeding birds. Currently there is very little information about this segment of the population, and without this information, it is not possible to estimate the continental population size with any degree of precision. Life-tables have not been established for any of the breeding populations, age of first breeding can vary, and year-to-year differences in non-breeding by adults can be significant (Hatch 1995; Hatch and Weseloh 1999). At various times, different investigators (e.g. Lewis 1929; McLeod and Bondar 1953; Price and Weseloh 1986; Watson et al. 1991) have estimated that a range of 1.0 – 4.0 non-breeding cormorants per breeding pair determines the size of the non-breeding segment of North American cormorant populations. In recent estimates of population size, these estimates have been applied to breeding numbers. Hatch (1995) estimated the continental breeding population at > 360,000 pairs from data obtained mainly in the 1980s and early 1990s; applying the range of estimates for non-breeders, he estimated a continental population size of 1 – 2 million birds. Tyson et al. (1999) estimated a minimum of 372,410 breeding pairs in the U.S. and Canada, based on data collected between 1975 – 1997 (most data were collected 1994 – 1997). Applying the range of estimates above for non-breeders, Tyson and colleagues conservatively estimated that > 1 million individual DCCOs currently occur in the U.S. and Canada. Hatch (1995) noted that totals are imprecise because large regions have been poorly studied and because some of the largest populations are the least well known (e.g., Manitoba, Mexico). Additionally, review of the estimates by Lewis (1929), McLeod and Bondar (1953) and Price and Weseloh (1986) indicated estimates for non-breeders were not based on detailed scientific study and rigorous data. Before estimates for this segment can be made with any confidence, careful study and observations at specific colonies (e.g. older, stable colonies vs. newer, rapidly expanding ones), as well as at locations without colonies where non-breeders may aggregate, need to be undertaken (D.V. Weseloh, pers. comm.).

In addition to providing important information for total population size, an estimate of non-breeders is also important for predicting population growth, and effectiveness and impacts of potential management strategies. For example, at the large Oostvaardersplassen Great Cormorant (GRCO) colony in The Netherlands, Van Eerden and Van Rijn (1997) estimated that at least 40 % of birds present did not breed. Based on this estimate, they concluded that any intervention in colonies aimed at reducing number of breeders is likely to allow non-breeders to take over the empty territories, or disperse and colonize new territories.

For predicting appropriateness of management strategy, an estimate equally important to total population size (e.g., regional, continental) is the proportion of birds utilizing various habitats and resources. For example, Van Eerden and Van Rijn (1997) reported that fish farms provide food for only about 0.5 – 1 % of the total population of European GRCOs in winter, and thus managing on the local level was considered more appropriate. We do not have good estimates for the proportion of the DCCO population utilizing aquaculture farms. For example, Nisbet (1995) estimated that the number of DCCOs wintering in the Mississippi Valley is probably equivalent to no more than five percent of the total interior population. However, this estimate represents the number present at a given point in time rather than the total number of birds moving through the region, which may be a larger proportion of the total population. Dolbeer (1991) estimated from banding records that 120,000 birds might move through this region during winter months. J. Glahn (pers. comm.) suggested that the number currently passing through has probably more than doubled since Dolbeer's (1991) estimate. There are two levels of uncertainty surrounding these estimates. First, we cannot determine the proportion of the

population the migrants represent because total population size is unknown. Second, because it is not clear how many of these birds are overwintering and how many are simply passing through, it is difficult to determine level of impact this portion of the population has on aquaculture. Therefore, better estimates on the proportion of the population utilizing fish ponds and the variation of residence time among birds present in the Mississippi Delta region are important for decisions about whether to manage cormorants on a population or local level (see PART V. MANAGEMENT OPTIONS).

Figure 1. Distribution of Double-crested Cormorant (*Phalacrocorax auritus*) breeding colonies in North America 1970 – 2000.

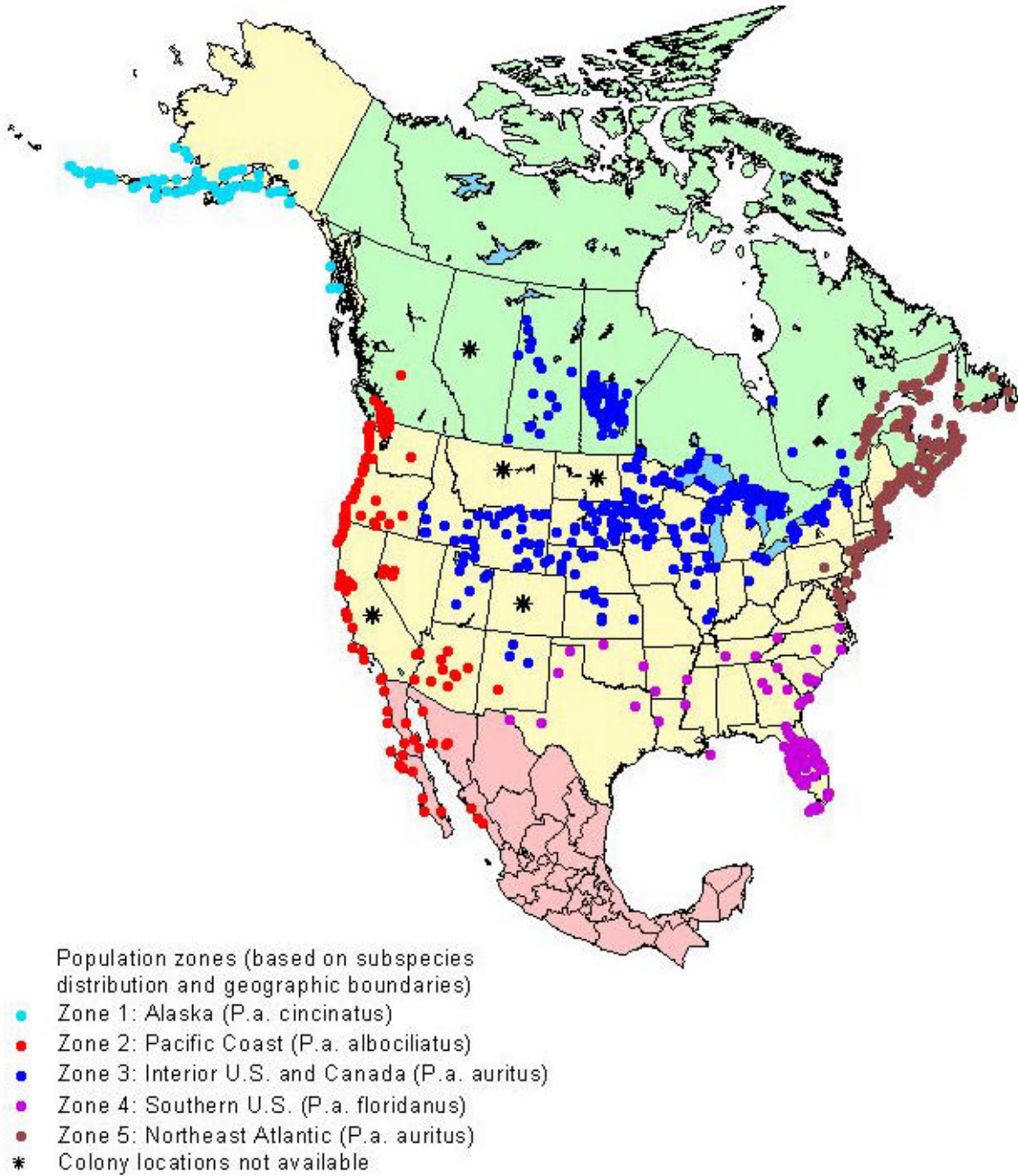
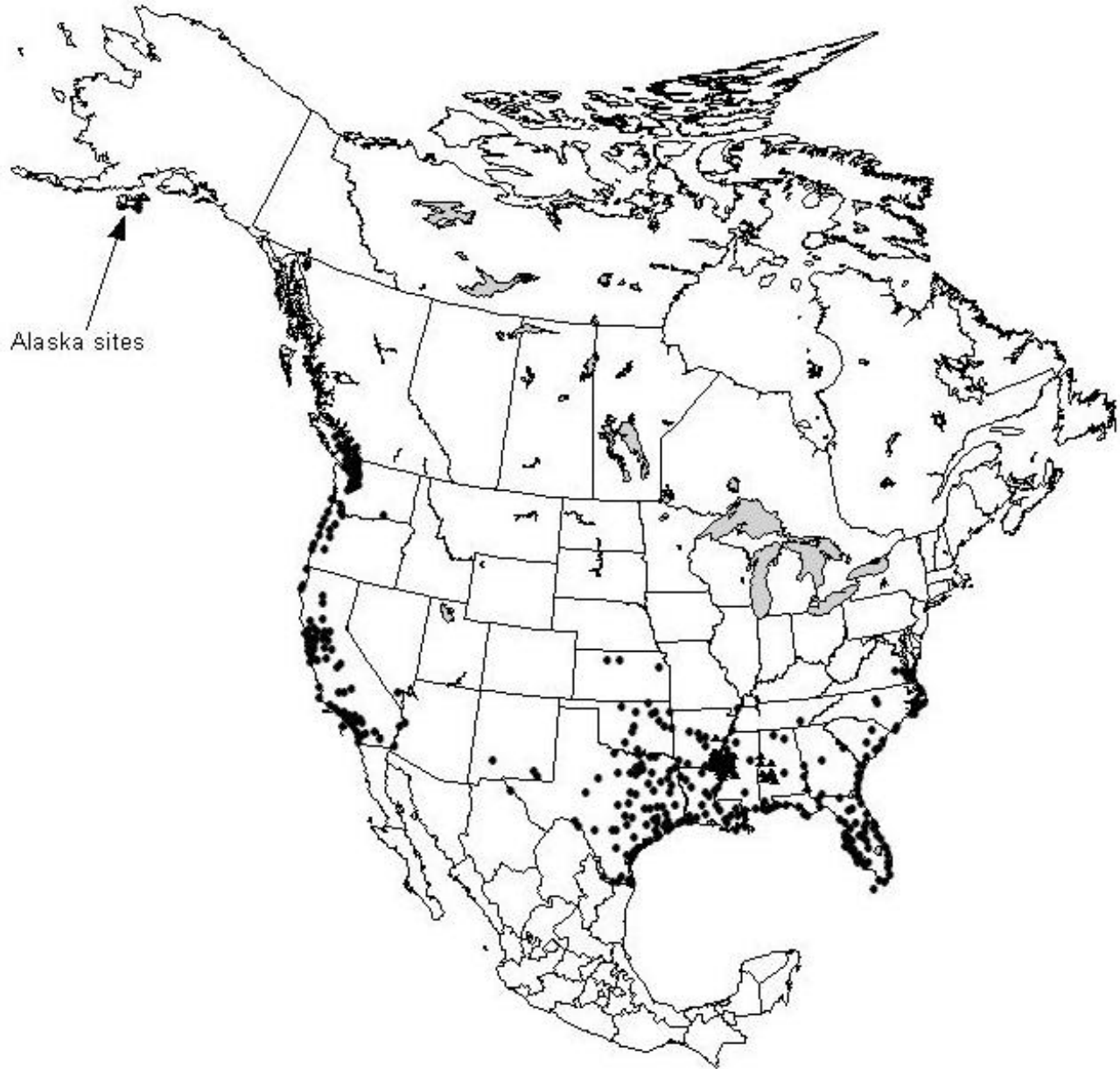


Figure 2. DCCO Christmas Bird Count sites (1984 – 1998) where ≥ 100 birds were counted at least once.



ZONE 1: SUMMARY OF POPULATION DATA FOR ALASKA

Introduction

In Zone 1, the breeding population occurs entirely in Alaska, at Nunivak Island and the southeast Bering Sea, and from the Aleutian Islands to the southeast coast, including Kodiak Island. Inland it occurs to Lake Louise; the first nest in the Yukon Territory was confirmed at Lake Laberge in 1998 (Hatch and Weseloh 1999). The westernmost limit occurs in the eastern Aleutian Islands at Chuginadak Island (52°51'01" N, 169°49'41" W) (Carter et al. 1995). Because DCCOs breeding in Alaska represent the entire *P.a. cincinatus* subspecies (Carter et al. 1995), the largest of the five subspecies (Hatch and Weseloh 1999), the summary of this population zone is fully treated in the Alaska state summary, below.

Alaska

Summary of Population Data and trends

(Surveys completed by Don Dragoo, Alaska Maritime NWR, USFWS, Homer; and Shawn Stephenson, Migratory Bird Management, USFWS, Anchorage.)

Breeding.

While we did not determine when DCCOs were first documented as a nesting species in Alaska, it is clear that the species has been a long time breeder in the state (see Historical Information, below). Between 1970 and 2000, the species was confirmed breeding at 126 colonies. Most breeding occurs along the southern coast and on the Aleutian Islands (Figure 4). Because Alaska has not conducted statewide censuses of all colonies in the same year, the number of breeding birds is not known. Additionally, most colonies have only been censused once or not since the 1970s (Carter et al. 1992); therefore population trends are not available. However, of 126 known colonies, 106 were counted at least once between 1970 and 2000; the vast majority (93 %) were small, with < 100 pairs on average. Carter et al. (1995) estimated a total of 2,811 pairs bred at 90 coastal colonies between 1970 and 1992, at which time comprised 12 % of the Pacific Coast population, defined by Carter et al. (1995) to encompass *P.a. cincinatus* in Alaska, and *P.a. albociliatus*, breeding from southern British Columbia to Sinaloa, Mexico. In the interior, colonies have not been adequately surveyed.

Numbers are thought to have declined since historical times, especially after introduction of predators (see Historical Information, below). At present, oil spills threaten cormorants in several areas, and human disturbance at breeding colonies remains a problem (Carter et al. 1995). On coastal islands, introduced predators [e.g., red and arctic fox, Norway rat, ground squirrel (*Spermophilus undulates*), rabbit are being removed], which may enable former breeding sites to be re-colonized by cormorants (Bailey 1993; Carter et al. 1995).

Winter.

In winter DCCOs occur, but no information was provided on wintering locations and winter censuses are not conducted. Gabrielson and Lincoln (1959) reported the species was resident in its breeding range, except possibly in the more northern areas, and that it was a fairly common winter bird in southeastern Alaska. During CBCs conducted between 1984 – 1998, only 3 sites had ≥ 100 DCCOs during at least one count, and fairly small numbers were observed

overall (Figure 3). The CBC data have not been analyzed by the Patuxent Wildlife Research Center for trends during this period, and no trends are readily apparent from Figure 3.

Migration.

No information was available on migrants or migration in Alaska.

Historical Information

The DCCO is a long time resident and breeding species in Alaska. Remains of DCCOs found in middens on Amchitka Island date back 2,650 years (Siegel-Causey et al. 1991). Midden remains indicate DCCOs were formerly more abundant in the central Aleutian Islands, and climate changes are thought to have probably reduced numbers in Alaska over time (Carter et al. 1995). While it has been suggested that the introduction of arctic foxes onto nesting islands beginning in 1750 (Bailey 1993) probably greatly reduced breeding numbers (Carter et al. 1995), it is not entirely clear how severely these introductions impacted DCCOs. In the late 1800s, DCCOs were reported as abundant residents and breeders in the Near Islands in the western Aleutians (Turner 1885), where foxes at this time were already present; thus whatever damage foxes had done would already have been evident (V. Byrd, pers. comm.).

Land Ownership

No information provided.

Productivity

In 1998, productivity was monitored at Aiktak Island, and 1.8 chicks / nest fledged (Byrd et al. 1999). On Duck Island, productivity was monitored in both 1998 and 1999, and was 0.0 and 0.13, respectively (Byrd et al. 1999; D. Dragoo, pers. comm.).

Figure 3. December estimates of DCCOs in Alaska based on CBCs, 1984 – 1998.

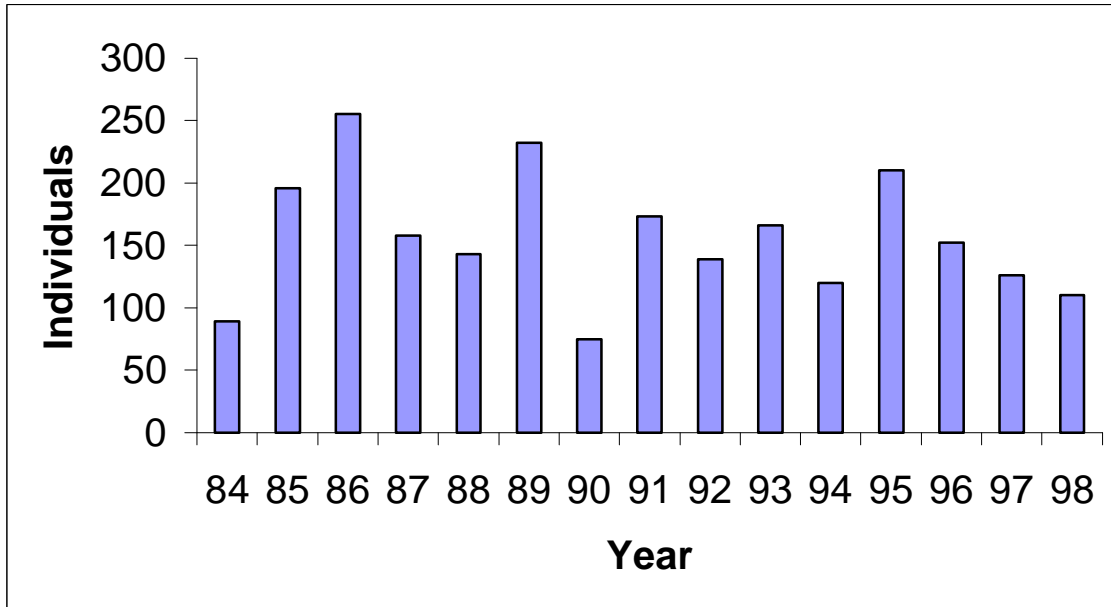
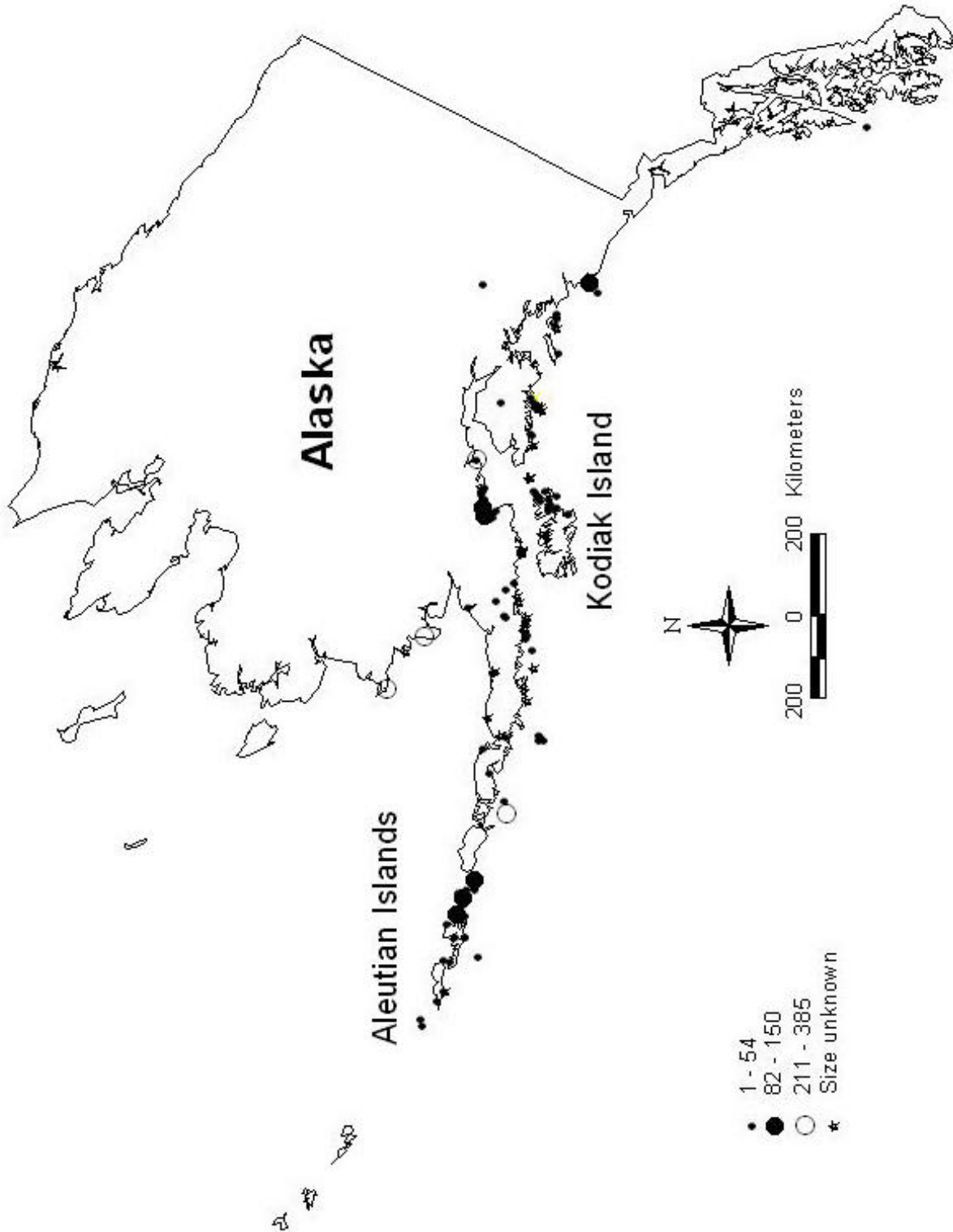


Figure 4. Distribution and size of active colonies in Alaska at time of last surveys (1970 – 1999).



ZONE 2: SUMMARY OF POPULATION DATA FOR THE PACIFIC COAST

Introduction

Breeding Information

In this zone, the breeding population occurs mostly along the coast from southern British Columbia south to at least Bird Island, Sinaloa, Mexico; some birds may nest farther south (Carter et al. 1995; Hatch and Weseloh 1999). Birds in this zone comprise the *P.a. albociliatus* subspecies (Baird et al. 1884; Palmer 1962). Though most nesting occurs along the coast, significant colonies also occur or have occurred inland in Washington, Oregon, California and Mexico. Additionally, most birds breeding in Nevada, Arizona and New Mexico are presumed to be members of this subspecies. However, Johnsgard (1993) suggested that birds breeding in western Nevada at Pyramid Lake would seem to be more geographically affiliated with *auritus*, but we have chosen to group the Nevada birds in population Zone 2 based on geographic proximity to California and Arizona birds. In New Mexico, most of the breeding birds are probably *P.a. albociliatus*, but a small number likely represent *auritus* (S. Williams, pers. comm.). At this time we decided to include New Mexico birds in the Pacific Coast population because *auritus* numbers appear to be marginal and data are still being analyzed (see New Mexico profile for further detail). Birds breeding in Idaho and Utah have characters intermediate between *P.a. albociliatus* and *auritus*, but these birds have been grouped in Population Zone 3 (see summary for Zone 3: Summary of Population Data for Canadian and U.S. Interior Population for further information). Table 1 shows the status (Breeding, Wintering, Migrant) of the Double-crested Cormorant throughout Zone 2.

No survey for this assessment was completed by several states in this region because breeding data for much of the Pacific Coast have been compiled into one database by the USFWS, Office of Migratory Birds and Habitat Programs-Pacific Region. We used this database to acquire information on cormorant distribution and abundance.

For many states and provinces in this zone and Alaska, we cite percent of the Pacific Coast population that breeding pairs in a particular area comprised, based on data from Carter et al. (1995). They define the Pacific Coast population to encompass *P.a. cincinatus* in Alaska, and *P.a. albociliatus*, breeding from southern British Columbia to Sinaloa, Mexico. Therefore, when we give estimates of percent of the Pacific Coast population that pairs from a particular area comprised, we refer to the Pacific Coast population as defined by Carter et al. (1995). Though their definition of the Pacific Coast breeding population differs from ours, we cite Carter et al.'s (1995) percent estimates to provide additional information on the number of birds occurring within this region.

Along the coast, from British Columbia to Mexico, Carter et al. (1995) estimated that approximately 21,849 pairs nested between 1968-1992. More recent estimates for the entire region are not available, but significant changes occurred during the 1990s. Large increases were documented along the coast in Oregon at the mouth of the Columbia River, and inland in California at the Salton Sea; increases in these two areas account for most of the growth in the region. Sharp declines were observed along coastal British Columbia and Washington colonies.

Because data are not available for birds breeding in significant portions of this region (e.g., Mexico, some interior areas) and data collection has not been coordinated at a regional scale, it is not possible to summarize trends for the population as a whole. Additionally, trends in this region are strongly affected by apparent movements of birds during El Nino oceanographic

conditions, habitat loss at interior colonies, and use of artificial habitat in some areas (Carter et al. 1995). With these habitat changes, distributional changes appear to be occurring; large numbers of immigrants may account for much of the growth seen in particular areas (see California and Mexico profiles). How these distributional changes affect overall population growth for Zone 2 is not clear.

Winter and Migrant Birds

Though portions of the Pacific Coast and the interior provide wintering habitat for DCCOs breeding in this region, no state- or province-wide counts of wintering DCCOs have been conducted by state or provincial agencies monitoring cormorant numbers. Therefore, with the exception of British Columbia, which reported that all breeding DCCOs are residential, no precise estimates are available for winter numbers in this region. Additionally, while many of the birds that breed in this zone are year round residents, some migration does occur, and it is difficult to distinguish migrants from residents. Overall estimates of migrants in the region have not been obtained.

Analysis of recent (1989 – 1998) CBCs conducted in this region has not been undertaken, and trends are not readily apparent from data collected during counts conducted between 1984 – 1998 (Sauer et al. 1996) (Figure 5). However, CBCs indicate both coastal and interior areas of California provide major winter habitat; numbers from California counts are much larger than those reported in any of the other states (but see California profile for limitations of this data). CBC data also suggest that coastal areas in British Columbia, Washington and Oregon provide winter habitat for significant numbers of birds. In the interior, significant numbers are reported in Arizona on larger reservoirs along the Colorado River, especially near Yuma, and in New Mexico in the Elephant / Butte Caballo and Carlsbad areas (Figure 2).

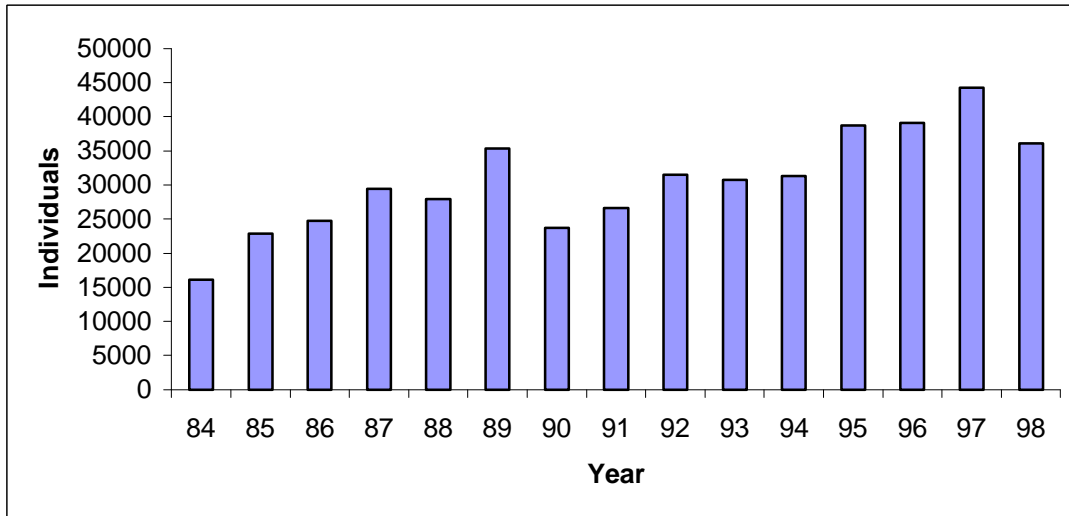
Several areas in Mexico also provide significant winter habitat. The DCCO is commonly found along both coasts of Baja Sur, and along the Gulf Coast of Tamaulipas and Campeche. It is also a fairly common to common resident from the northwest Yucatan to northern Belize (Howell and Webb 1995). However, no information is available on winter numbers in Mexico.

Historical Information

Carter et al. (1995) reviewed the historical background of the DCCO on the Pacific Coast, and that information is summarized here. As in many parts of its range, the DCCO experienced substantial decline and loss of breeding colonies along several portions of the Pacific Coast in the 1800s and early 1900s. Human activities resulted in habitat disturbance and occupation, and breeding birds were directly persecuted by humans at their nesting colonies, where they were shot and their nests were destroyed. In the interior, agricultural and water developments replaced much important habitat. The decline in numbers is thought to have been more pronounced than the available sparse literature indicates, and probably resulted in the complete loss of DCCO colonies in regions first settled by Europeans (e.g., southern British Columbia, Puget Sound, Columbia River, San Francisco Bay). While small populations did survive, reproductive success (through DDT and other pesticide-caused eggshell thinning, see Carter et al. 1995) and population growth was suppressed by environmental contaminants. However, after several decades of reduced human persecution and disturbance, legislation introduced in the 1970s to ban DDT and regulate pesticide use, and addition of the DCCO to the Migratory Bird Treaty Act Protected List (1972), the DCCO began to recover and expand in this region. The source for this expansion was probably the remnant breeding groups that had

persisted in several areas (e.g., Olympic Peninsula, South Farallon Islands, Channel Islands, Mexico). Nevertheless, habitat in this region has been extensively changed by humans, and as a consequence DCCOs still show local declines in certain areas. Additionally, records from the late 1800s and early 1900s indicate that DCCOs existed in much larger numbers than they do today in many areas of this region (e.g., see California and Mexico profiles).

Figure 5. December estimates of DCCOs on the Pacific Coast, Zone 2^{*}, based on CBCs, 1984 – 1998.



* States and provinces included: AZ, BC, CA, NV, NM, OR, WA; annual totals based on sites with ≥ 100 individuals.

Table 1. Zone 2, Pacific Coast. Summary of Population data and trends.

(+ = increasing; - = decreasing; 0 = no change).

State/ Province	Status (B,M,W, w) ¹	Number colony sites/time period	Active colonies/last year surveyed	Number of pairs/last year surveyed	Breeding trend	Winter number (birds)	Migrant number (birds)
AZ	B, M, W	12/1968-99	10/1993-99	NA	NA	1000s?	NA
BC	B, W	18/1070-99	14/1983-91	800/1999	-	1600	-
CA	B, M(?), W	45 coastal; ≤55 interior 1970-99	39/1989-91 coastal; 32/1999 interior	5,092/1989- 91 coastal; 6,900/1999 interior	+	1000s, X = 20,945 (10,688- 35,657)	NA
MX	B, M(?), W	27/1968-92	27/1968-92	7,269/1968- 92	NA	NA	NA
NM ¹	B, M(?), W	2/1996 ²	2/1996 ²	600/1996	-/?	2,000+ ³	NA
NV	B, M, W	5/1986- 2000	5/1986-1999	NA	-/?	10-100s?	1000s
OR	B, M, W	40/1979- 2000	30/1988-98	6,249/1988- 92	+	1000s, X = 1,548 (658- 2,746)	NA
WA	B, M(?), W	43/1970-99	22/1992 coastal; 1/1997 interior	1,618/1992 coastal; 652/1997 interior	-/?	1000s, X=4228 (2250- 6821)	NA

1 Lower Rio Grande colonies; excludes birds in Zone 3 (Table 4)

2 Multiple sites at Elephant Butte and Caballo Reservoirs

3 Did not distinguish *P.a. albocilatus* from *auritus*.

Figure 6. Distribution of Pacific Coast DCCO breeding population (Zone 2), 1970 – 2000.

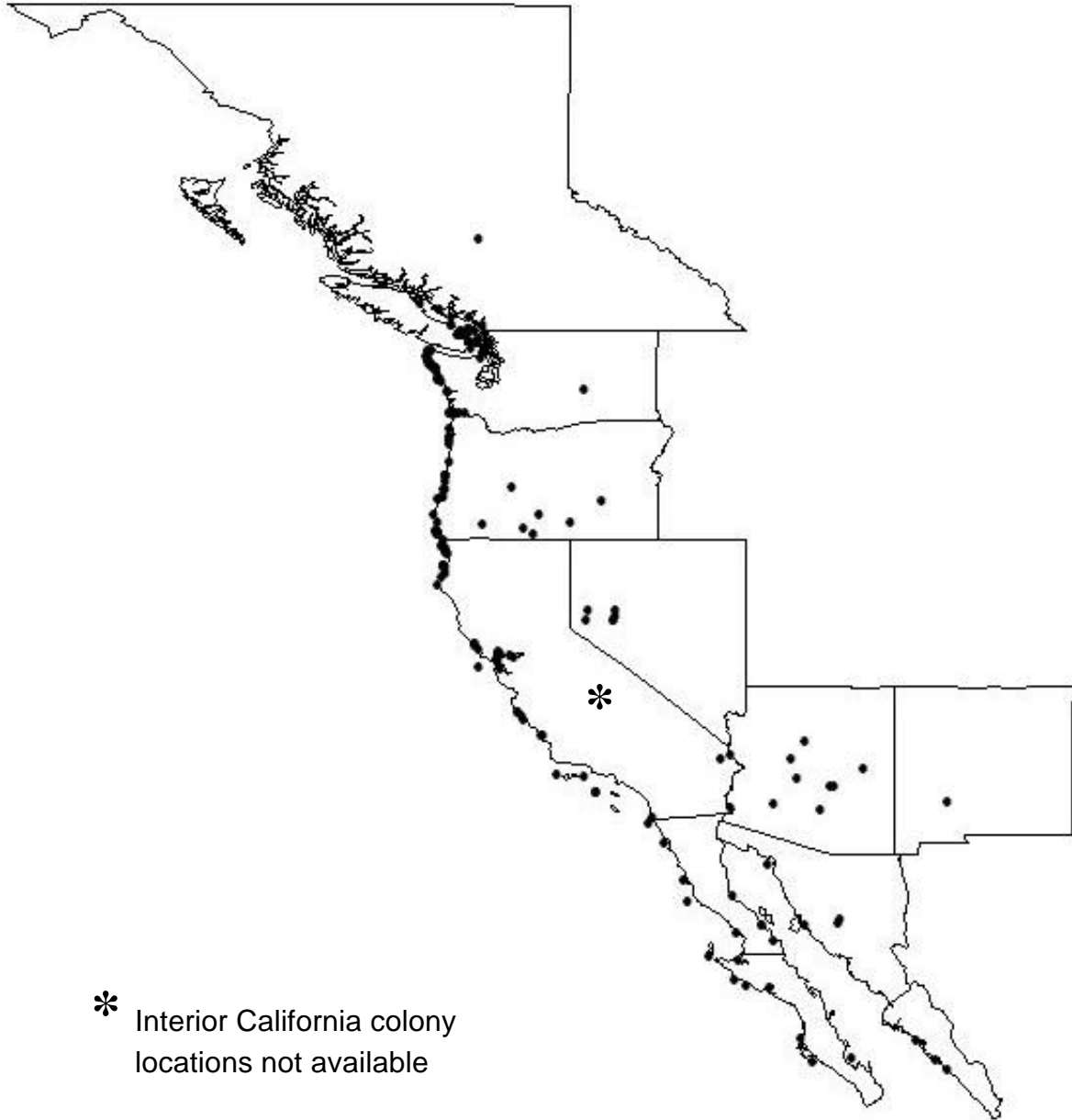
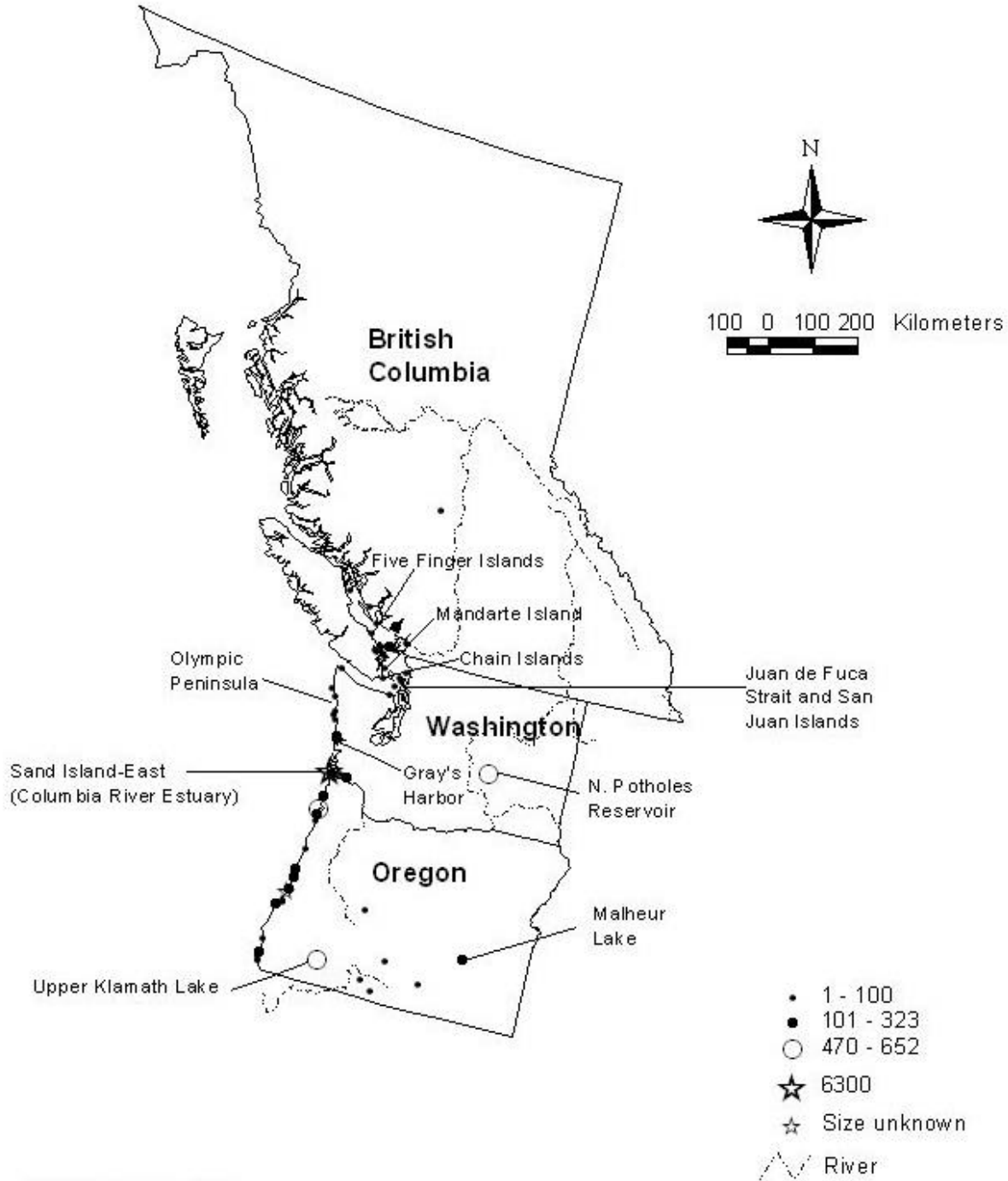


Figure 7. Distribution and size of active colonies on north Pacific Coast at time of last surveys (1990s)¹



¹ 87% counted in 1990s with one colony last counted in 1970s.

Figure 8. Distribution and size of active colonies on south Pacific Coast at time of last surveys (CA = 1989 – 1998; MX = 1973 – 1992).

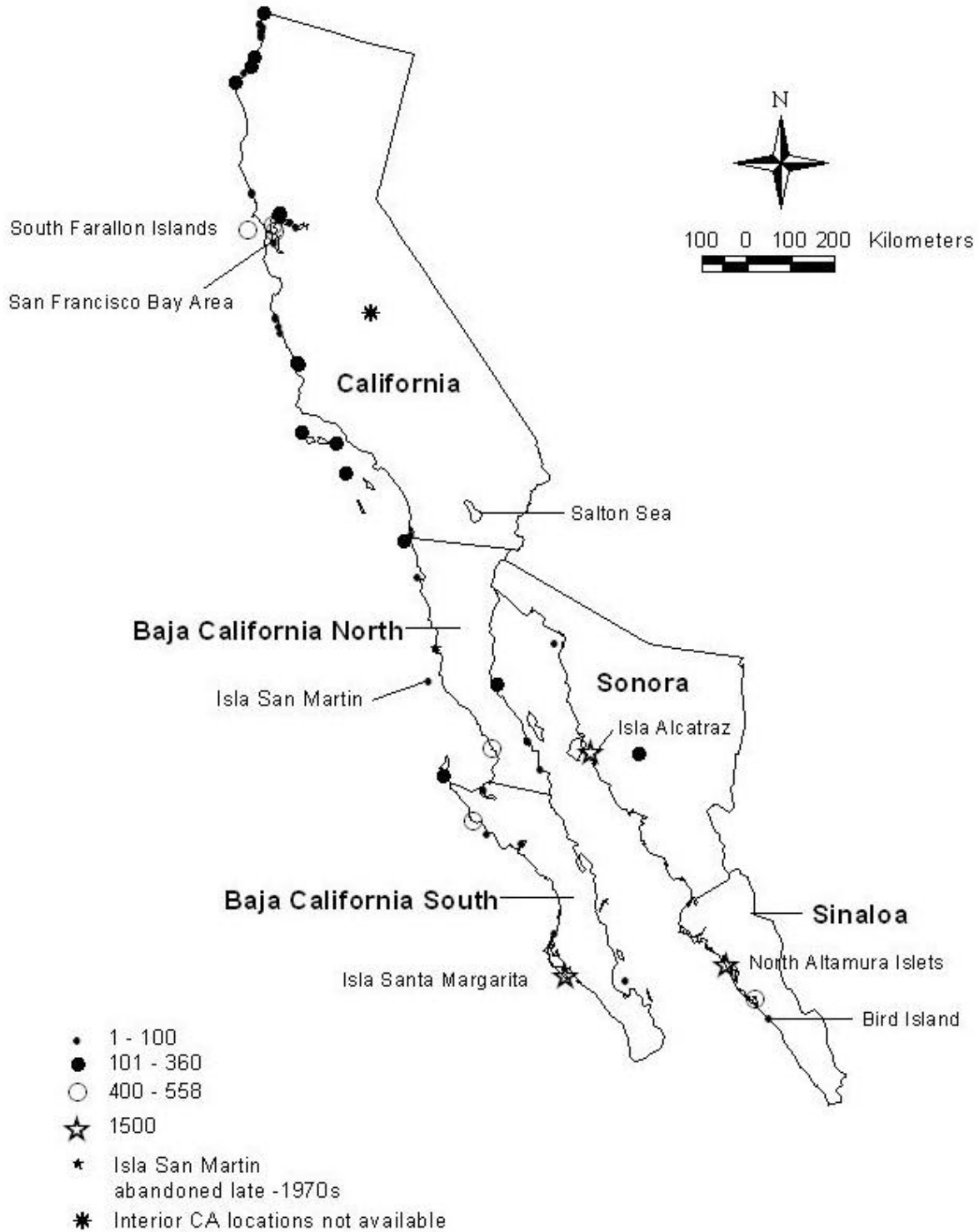
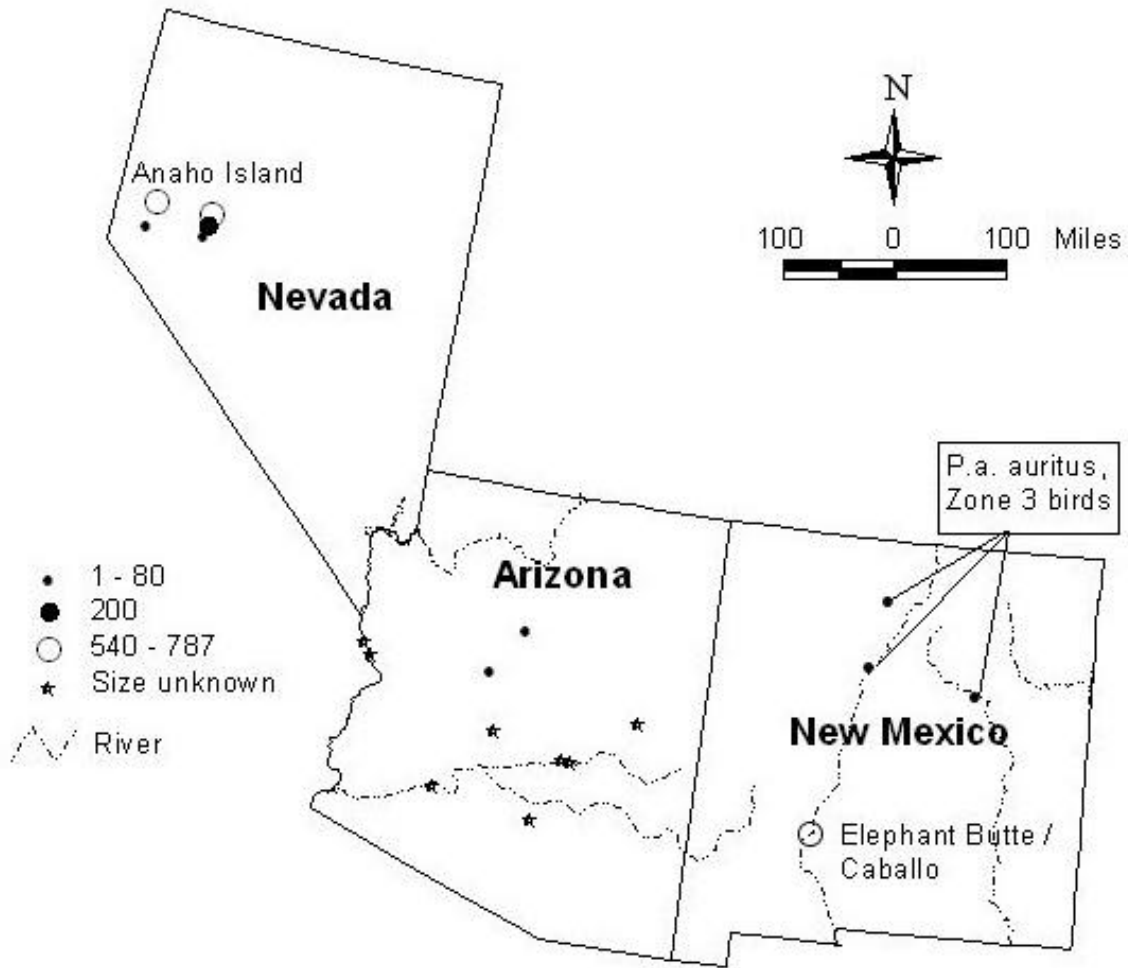


Figure 9. Distribution and size of active colonies in the Pacific Coast southwestern states (1993 – 2000).



Arizona

Summary of Population Data and Trends

(Survey completed by Troy Corman, Arizona Game and Fish Department.)

Breeding.

We did not determine when the DCCO was first documented as a nesting species in Arizona (see Historical Information, below); currently there are 12 known colony sites. Of these, 10 were active during the Arizona Breeding Bird Atlas Survey, conducted between 1993 – 1999 (Arizona Breeding Bird Atlas, unpubl. data). The other two colony sites were documented prior to the Atlas survey, one at Imperial NWR, where the species was reported breeding between 1975 – 1984 (Rosenberg et al. 1991), and the other at the San Carlos Reservoir, where breeding was documented in 1968 (Monson and Phillips 1981). Because DCCOs are not monitored in Arizona and regular surveys are not conducted, information on trends is not available. DCCOs in Arizona are presumed to be *P.a. albociliatus* (Baird et al. 1884; Monson and Phillips 1981).

Winter.

In winter, the DCCO occurs throughout southern and western Arizona on all lakes, rivers and canals. However, estimates of the number of overwintering birds are not available. Concentrations > 500 are probable only at larger reservoirs along the Colorado River, especially near Yuma. During Christmas Bird Counts conducted between 1984 – 1998, only 3 sites had \geq 100 DCCOs during at least one count (Figure 10). The majority of birds were counted at Martinez Lake-Yuma, where a high count of 3,092 birds was recorded in 1992. While the CBC data have not been analyzed by the Patuxent Wildlife Research Center for this period, Figure 10 suggests that numbers have declined since 1992. In the last 3 counts included here, conducted between 1995 – 1998, numbers observed were the lowest recorded during the 15 year period examined.

Migration.

During spring migration, peak numbers occur late March – April; in fall, peak numbers are observed October – November. No information was available on peak numbers, and locations of major stopover sites were not provided.

Historical Information

Little historical information was obtained. The species was described as “rather common” around Tucson from March 29 to June 20, 1897 (Phillips et al. 1964). Elliot Coues (1866) mentions that it occurred on the lower Colorado River. Date of first documented breeding was likely recorded between 1910 and 1964. In 1910 the California Museum of Vertebrate Zoology conducted an expedition down the Colorado River from April to May, and documented the species at various points along the river between the Laguna Dam and Yuma. *The Birds of Arizona* (Phillips et al. 1964), states that the DCCO breeds locally along the Colorado River (T. Corman, pers. comm.).

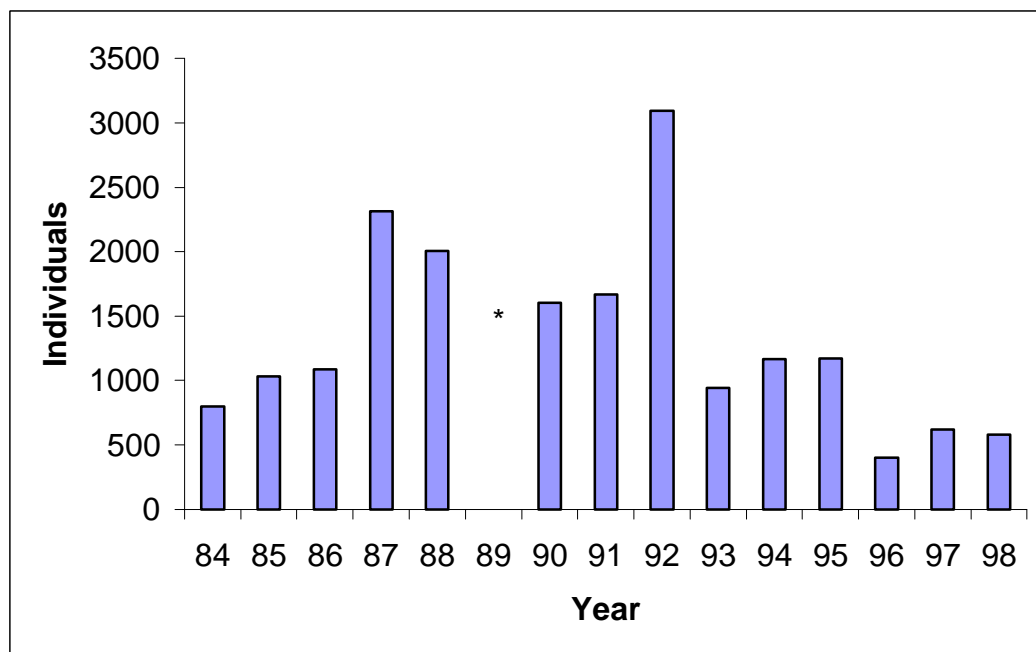
Land Ownership

No information provided.

Productivity

No estimates available.

Figure 10. December estimates of DCCOs in Arizona based on CBCs, 1984 – 1998.



* No data available for 1989.



British Columbia

Summary of Population Data and Trends

(Survey completed by M. J. Chutter, B.C. Min. Env., Lands and Parks, Wildlife Branch; I. Moul, Foul Bay Ecological Research Ltd., B.C., contributed breeding and trend data.)

Breeding.

The DCCO has been documented as a breeding species in British Columbia since 1927; however, archaeological evidence indicates the species occurred and may have been breeding in the area much earlier (see Historical Information, below). British Columbia represents the northernmost extension of the subspecies *P.a. albociliatus*, breeding as far north as Mitlenatch Island (49°57' N, 125° W). Between 1970 and 2000, a total of 19 breeding sites was recorded, 18 along the coast and 1 in the interior. (Figure 7). Along the coast most colonies occur in two main regions, the Northern Strait of Georgia and the Gulf Islands; most breeding occurs in the latter region. Carter et al. (1995) reported that 1,751 pairs nested at 14 active colonies in southern British Columbia between 1983 – 1991, comprising 7 % of the Pacific coast marine population.

Currently, 602 pairs are estimated to breed along the coast (I. Moul, pers. comm.). In the interior, breeding was first documented in the mid-1990s in the central portion of the province at Stum Lake, where small numbers (6 – 12 pairs) have been reported (M. Chutter, pers. comm.).

Between 1935 – 1987, numbers of breeding birds increased in the province fairly steadily (Carter et al. 1995). However, in the 1990s, many colonies were abandoned and most had population declines. In recent years these declines became more dramatic. Between 1987 and 2000, provincial breeding numbers declined from 1,981 pairs to 602 pairs. In the Gulf Islands, counts from 13 colonies conducted in 2000 indicate a highly significant drop (70 %) in estimated number of nesting pairs since 1987. Numbers at the three largest Gulf Islands colonies (Great Chain, Mandarti and Five Finger Islands) have declined fairly steadily since the late 1980s. In 1987, these three colonies comprised about 82 % of the total pairs estimated in British Columbia, and about 94 % of the total pairs breeding in the Gulf Islands. Declines were observed beginning in 1990, with the sharpest declines occurring in the late 1990s (Figure 11). The province's largest colony, located at Mandarti Island, had nearly 1500 nests in 1981, but in 2000, only 215 nests were counted. Additionally, there appeared to be no successful reproduction at any of these colonies in 1998 and 1999 (data not available for 2000).

Because of these declines, the DCCO is designated “at risk” in the province and is on the provincial Blue List of species considered to be Vulnerable. Recent re-evaluation of the DCCO suggests that it may warrant upgrading to the province's Red List for consideration of Threatened Status (M. Chutter, pers. comm.).

It is not clear if declines in numbers are in response to an imbalance in the ecosystem, caused by human activities, or reflect some long term natural cycle. Moul (2000) reported repeated observations of Bald Eagles flying into the vicinity of colonies located along the southeast coast of Vancouver Island. In response to eagle arrival, the entire colonies would flush from nests. Gulls, and occasionally crows, often returned to nest sites quicker than cormorants and depredation of unattended cormorant eggs and chicks was observed. [Moul also observed similar responses to human caused disturbances, i.e. marine wildlife viewing expeditions getting close enough to flush birds (M. Chutter, pers. comm.).] This scenario happened at least once / day and sometimes several times per day until late June / early July (Moul 2000). Colony failure appeared linked to this phenomenon (Moul 2000). Human activities such as fishing, marine wildlife viewing or picnicking in close proximity to nesting cormorants can also contribute to colony declines and abandonment (Campbell et al. 1990; Carter et al. 1995; Moul pers. comm.; M. Chutter, pers. comm.). Presently, there is concern that the DCCO could soon be extirpated as a breeding species (Moul 2000).

Winter.

In winter, DCCOs are widely scattered in protected waters along the outer coast. They are most abundant in the vicinity of the Strait of Georgia and Juan de Fuca Strait, where the species is resident (Campbell et al. 1990; M. Chutter, pers. comm.). Based on the number of coastal breeders in this area, at least 1200 individuals overwintered in 2000. However, this is likely an underestimate. Campbell et al. (1990) notes the occurrence of seasonal movements of birds, most likely involving Alaskan breeders in spring and fall. Presumably some of these birds overwinter in BC, which would add to the province's overwintering resident population (M. Chutter, pers. comm.). There is also no information available on the number of nonbreeding and young birds that overwinter.

Campbell et al. (1990) reported wintering numbers counted on CBCs at Vancouver, Ladner and Victoria noticeably increased between 1958 and 1984; between 1960 and 1984, the

average wintering population at these three sites nearly tripled. During counts conducted at these three sites between 1984 – 1998, numbers fluctuated; in some years increases occurred but were then followed by declines (1987, 1991, 1992, 1996). This appears to be the general pattern for the province as a whole during this time period (Figure 12). During earlier counts conducted between 1959 – 1988 an increase of 3.6 % per year was observed (Sauer et al. 1996). More recent data have not been analyzed by the Patuxent Wildlife Research Center and information on trends is not available. CBCs conducted in 1998 estimated 2,390 birds in the province. Based on known breeding survey data and the assumption that all breeders are resident, M. Chutter (pers. comm.) reported that the winter population probably peaked in the late 1980s at a minimum of 4,000 – 5,000 birds. The high CBC for the province, 3,726 birds, was recorded in 1993 (about 45 % of these birds were counted at the three sites mentioned above).

Migration.

Interior breeding birds at Stum Lake are migratory. There are no known important migratory stopover sites (M. Chutter, pers. comm.).

Historical Information

Though the first nesting record was not obtained until 1927 at Mandarti Island, subsequent observations suggest that the colony at Ballingal Islets was probably established earlier than this (see Carter et al. 1995 for records). Furthermore, DCCO bones are abundant at archaeological sites throughout the Strait of Georgia and indicate DCCOs occupied this area for the past 5,000 years (Hobson and Driver 1989). Thus, fairly recent colonization of the province probably represents re-colonization following a previous extirpation (Carter et al. 1995; Campbell et al. 1990).

After the initial discovery of breeding, numbers grew slowly, and have varied dramatically between colonies. In 1946, only two colonies were known, with a total of 50 pairs. By 1960, four colonies were active, with a total of 150 pairs. In 1975, there were six known colonies, with a total of 671 pairs (see Campbell et al. 1990 for records). Most rapid growth occurred sometime between 1975 – 1988. A high of 2,032 pairs breeding at 15 coastal sites was recorded in 1988 (Rodway in press, cited in Campbell et al. 1990). These increases were then followed by significant overall declines in the 1990s (see above).

Land Ownership

In British Columbia > 90 % of the land base is owned by the Crown. With the possible exception of Bare Point (which could be on land owned by the local pulp mill), all known breeding sites are presumed to be on Crown Land (land owned by the province). Several of these sites have some form of protected status (M. Chutter, pers. comm.). Marine wintering sites are under the jurisdiction of the federal government. Fresh water areas are technically on provincial lands, though surrounding lands could be privately owned.

Productivity

Some productivity data, collected in the 1990s (Moul 2000), were available for select colonies. These data are summarized below (Table 2).

Table 2. Numbers of chicks fledged at DCCO colonies along the east coast of Vancouver Island (from Moul 2000).

Colony	Year	Active Nests	Sample size	Successful Nests	Chicks fledged
Chain Islands	1995	432	432	0	0
	1998	300	300	0	0
	1999	100	100	0	0
Mandarti	1992	280	101	56	127
	1994	403	403	2	6
	1995	288	288	80	?
	1998	178	178	0	0
	1999	43	43	0	0
Crofton	1991	78	78	38	91
	1992	74	74	71	181
	1999	83	83	58	165
Hudson Rocks	1991	67	54	54	131
	1992	30	30	0	0
	1995	15	015	0	0
Five Finger	1991	118	47	45	115
	1992	191	26	24	72

Figure 11. Changes in number of pairs at Great Chain, Five Finger and Mandarti Islands, Gulf Islands, B.C., 1987 – 2000.

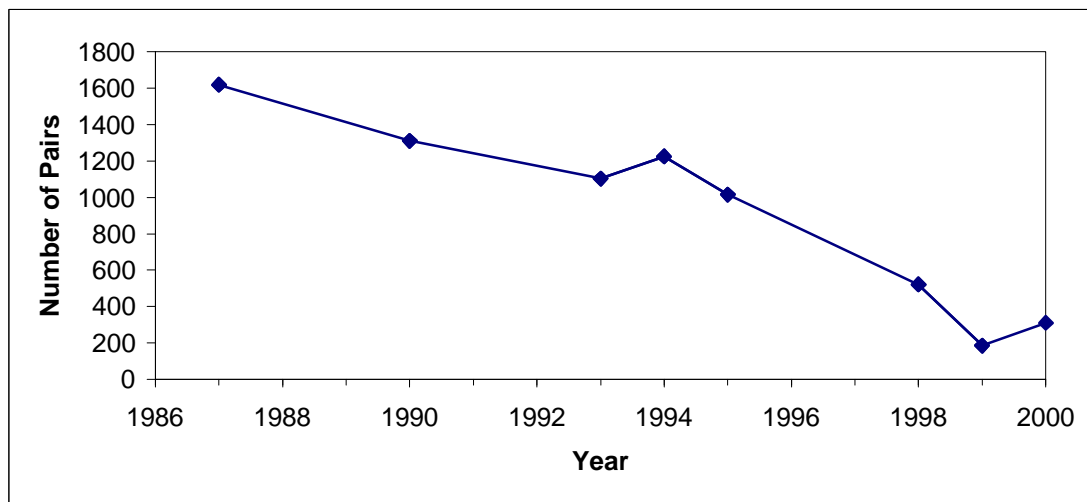
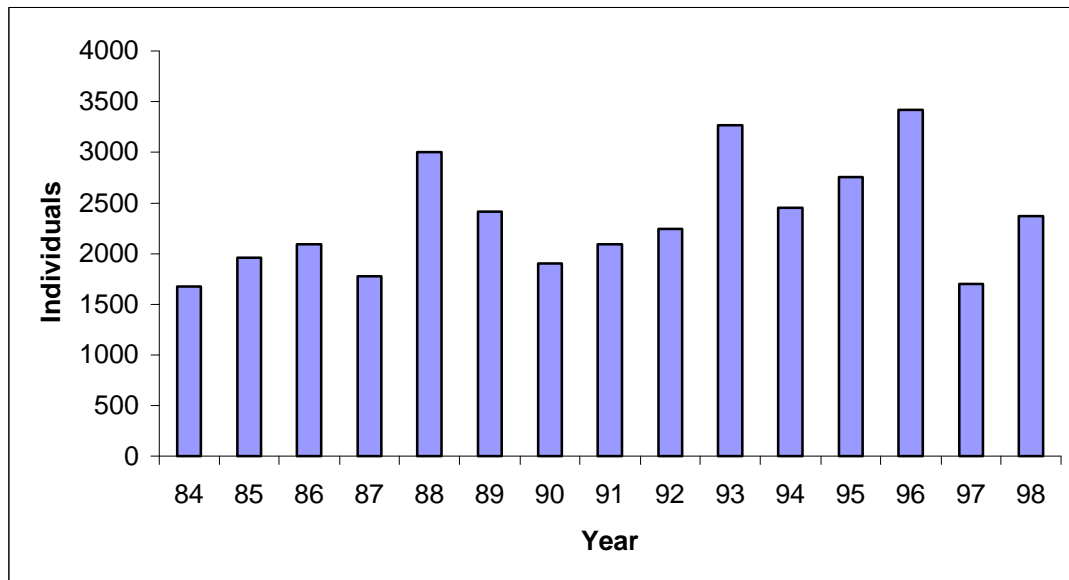


Figure 12. December estimates of DCCOs in British Columbia based on CBCs, 1984 – 1998.



California

Summary of Population Data and Trends

(No survey completed; breeding data collated by M. Naughton, Migratory Birds and Habitat Programs–USFWS, Pacific Region; and D. Shuford, Point Reyes Bird Observatory, CA.)

Breeding.

The DCCO has been a long time resident and breeding species in California (see Historical Information, below). Between 1970 and 1999 a total of 96 colonies was documented. Because interior colonies were not completely surveyed until 1997 – 1999, the total number of birds breeding in the state was not precisely known prior to 1997. Carter et al. (1995) reported results of surveys conducted by the USFWS and Humboldt State University between 1989 – 1991 for six breeding regions along the coast, where an estimated 5,092 pairs nested at 39 active colonies during this period, and comprised 21 % of the Pacific Coast marine population. Using aerial photography, complete surveys of coastal colonies have occurred annually from 1993 – 2000 by USFWS, U.S. Geological Survey, and Humboldt State University but data have not yet been obtained from photographs for all areas and years (Carter et al. 1996, 2000, unpubl. data). Complete interior surveys were first conducted between 1997 – 1999 by Point Reyes Bird

Observatory, and a total of about 6,900 pairs was estimated at 32 active colonies (D. Shuford, unpubl. data). The most recent data available for all regions are reported in Table 3.

For the coastal and interior regions, Carter et al. (1995) summarized trends up to 1992. In northern and central California, increases in coastal colonies were documented between the late 1960s and early 1970s up to late 1980s and early 1990s. In northern California, a new population emerged in the 1960s and has continued to expand. During 1969 – 1974, < 600 breeding birds were recorded between the Oregon border and Cape Mendocino; by 1989, numbers had increased to 3,252 birds, and several new colonies had formed. Along the south section of the northern coast and along parts of the central coast, declines occurred in the mid 1990s (Table 3). The colony at the South Farallon Islands is the only well documented breeding colony along the central coast. After being sharply reduced for nearly a century, increases were observed between the 1970s and 1980s, with colony peaks in 1982 and 1989. Though declines linked to major El Nino events occurred in 1983 and 1992, overall numbers of breeding cormorants increased from 1972 – 1995. However, data collected between 1990 and 2000 indicate this colony has been slowly declining over the last decade (Ainley and Boekelheide 1990; Carter et al. 1995, 1996, 2000; Abraham et al. 2000). In San Francisco Bay, DCCOs were common in the late 1800s and early 1900s, but were not recorded nesting in the Bay area until 1978 when they nested at Russ Island. By 1990, nine colonies were found on artificial structures or human-altered areas. More recent data for this area are available (M. Naughton, pers. comm.) but were not included in this assessment.

In southern California, numbers were very low in the early 1970s due to impacts from pollutants, oil spills (especially the 1969 Santa Barbara oil spill), and human disturbance (Gress et al. 1973; Gress 1994; Carter et al. 1995). DDT was banned around this time, and DCCOs began increasing shortly after 1975. Numbers in this area went from 208 pairs in 1975 – 1977 to 1,264 pairs in 1991, a growth rate of 12 – 14 %. This growth rate could not be explained by productivity alone and probably reflected immigration of birds from Mexico or interior California (Carter et al. 1995). Between 1991 – 2000, H. Carter (pers. comm.) reported numbers in southern California to have declined. Possible causes likely include: major El Nino events in 1992 – 1993 and 1997 – 1998; continuing pollutant issues (DDT and PCB); high levels of hooking by recreational fishermen in certain years; some deaths in gill nets and oil spills, especially the 1990 American Trader oil spill; and perhaps Newcastle's disease (H. Carter, pers. comm.).

In interior regions, Carter et al. (1995) reported only 1,403 pairs were known to nest at 12 active colonies by 1992. DCCOs had been decimated in these areas in the 1800s due to habitat destruction, loss of large lakes, and colony disturbances due to water and agricultural developments. However, a major increase in nesting occurred in the 1990s. By 1999, DCCOs nested at multiple locations. In 1999, about 80 % of all pairs occurred at the Salton Sea, where dramatic growth began occurring in the mid – late 1990s. Prior to this time period, DCCOs nested at the Salton Sea irregularly or in much smaller numbers. Between 1981 – 1995 at least partial counts / surveys were conducted each year. No more than 75 pairs were estimated in any count, and in many years no DCCOs were recorded (Salton Sea NWR files). In 1996, explosive population growth began coincident with colonization of Mullet Island at the south end of the Sea, and the population on Mullet Island reached 5,425 pairs in 1999 (Shuford et al. 2000). During this year, PRBO conducted a year-long reconnaissance survey of birds at the Salton Sea. As part of this work, four aerial photographic surveys of the nesting colony at Mullet Island (1 Feb, 19 Feb, 25 Mar, 16 April) were conducted. Cormorants on the sea or roosting along the

shoreline were counted by airplane, and during the breeding season the counts of cormorants at Mullet Island were added to the aerial surveys to provide grand totals of cormorants. These totals were 18,504 (early Feb), 11,160 (mid-Apr), 3023 (mid-Aug), and 15,179 (mid-Nov). Seven other colonies were active at the Salton Sea (largest 106 pairs) in 1999, but many of these may represent birds that failed at first attempts to nest at Mullet Island (Shuford et al. 2000).

The 1999 estimate for the Mullet Island Salton Sea breeding colony alone is essentially equivalent to the 1989 – 1991 estimate reported by Carter et al. (1995) for the entire coast. Similar to the growth observed in southern California between the mid-1970s to 1991, growth at the Salton Sea likely can not be explained by productivity rates alone; most of these birds are believed to be immigrants from unknown locations, possibly Mexico (D. Shuford, pers. comm.). Correspondingly large increases in winter numbers have also been observed here in the late 1990s (see below).

Winter.

The DCCO is mostly resident in California, and in winter large numbers occur over much of the breeding range. The species is observed throughout the near coastal counties and the Central Valley, along the lower Colorado River (Hatch and Weseloh 1999), and is fairly common at the Salton Sea year round (Garrett and Dunn 1981). CBCs conducted between 1959 – 1988 suggested an increase of 4.5 % per year (Sauer et al., 1996). In CBCs conducted between 1984 – 1998, significant numbers of DCCOs (≥ 100 birds in at least one year) were recorded at 65 sites, and a high of 35,657 birds was counted for the state in 1997. Though numbers appear to have increased in the late 1990s (Figure 13), the CBC data may be misleading and require cautious interpretation for a number of reasons. The recent data have not been analyzed for trends by the Patuxent Wildlife Research Center, and the number of CBCs conducted in California has increased during this period. Additionally dramatic increases have been observed at the Salton Sea during CBCs conducted in the mid-late 1990s (Figure 14). However, DCCOs start nest building at this site in December; therefore, CBCs at this site are tracking some breeding numbers. The recent large increases at the Salton Sea may mask any trend for the state as a whole (D. Shuford, pers. comm.). Large numbers also winter around Oakland, and an unusually high CBC of 6,781 birds was recorded in 1997.

Migration.

No information was obtained.

Historical Information

Breeding records obtained in the late 1800s and early 1900s are available for coastal and interior locations (Grinnell and Miller 1944; Carter et al. 1995). However, remains of cormorant chicks found in middens suggest breeding occurred prior to the arrival of Spanish explorers in the 1770s (Sher 1994 *in* Carter et al. 1995). The fact that bones were retrieved from middens indicates that native harvest of birds was occurring at the colonies. The colony at the South Farallon Islands is well documented, and in the mid 1800s was probably one of the largest colonies on the coast, as it was reported to have thousands of nesting DCCOs. Numbers here declined in the late 1800s due to human disturbance and remained low until the early 1970s, when increases began to be recorded, but numbers have not reached their former size (Ainley and Lewis 1974; Ainley and Boekelheide 1990; Carter et al. 1995).

Overall increases that began in the 1970s are probably best described as recovery rather than expansion or growth, because they occurred as DDT and other pesticides became less widely used and reproductive success improved (M. Naughton, pers. comm.).

Land Ownership

Coastal colonies are owned by: National Park Service (Channel Islands and Redwood National Parks), U.S. Bureau of Land Management (California Islands Wildlife Sanctuary), USFWS (Farallon Islands), Tolowa tribe (Prince Island), and others. Many colonies in the San Francisco Bay area occur on artificial structures (e.g., bridges) managed by various state and federal agencies.

Productivity

Stenzel et al. (1995) reported fledging rates for the colony at the Richmond-San Rafael Bridge in the San Francisco Bay estuary of 0.98, 1.78 and 1.70 chicks per nest between 1988 – 1990, respectively; in the South Farallon Islands colony, 35 km west of the mouth of San Francisco Bay, they reported fledging rates of 1.29, 1.13 and 0.76 during the same years. (Also see Ayers 1975; Gress et al. 1973, 1995; Lewis and Gress 1988; Ainley and Boekelheide 1990; Ingram 1992; Gress 1994; Ingram and Carter 1997; and Martin and Sydeman 1998 for productivity at coastal colonies in California.)

Table 3. Summary of most recent census data for California breeding regions.

Region (N = known number of colonies, 1970-99)	N = number of colonies for which data were available; n = number of colonies active				
	1989	1990	1991	1995	1999
Northern Coast – North Section (N = 17)	1,408 pairs ¹ (N = 15; n = 14)				
Northern Coast – South Section (N = 4)	218 pairs ¹ (N = 4; n=2)			75 pairs ² (N = 3 n ≥3)	
Central Coast – Outer coast North (N = 3)	570 pairs ¹ (N = 3; n = 1)			413 pairs ² (N = 3; n = 1)	
Central Coast – San Francisco Bay (N = 9)		1,429 pairs ¹ (N = 9; n = 9)			
Central Coast – Outer Coast South (N = 6)	198 pairs ¹ (N = 6; n = 6)				
Southern Coast (N = 6)			1,264 pairs ¹ (N = 6; n = 6)		
Interior (N ≤ 55)					6,900 pairs ³ (N = 37; n = 32)

1 = Carter et al. 1995

2 = Carter et al. 1996, 2000
 3 = Point Reyes Bird Observatory, unpublished data.

Figure 13. December estimates of DCCOs in California based on CBCs, 1984 – 1998.

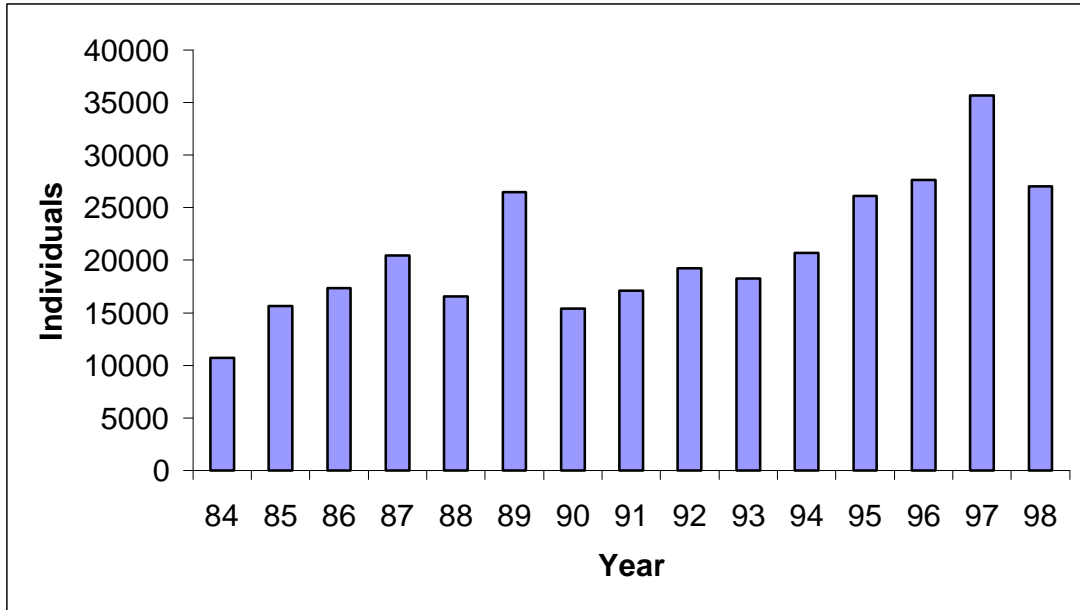
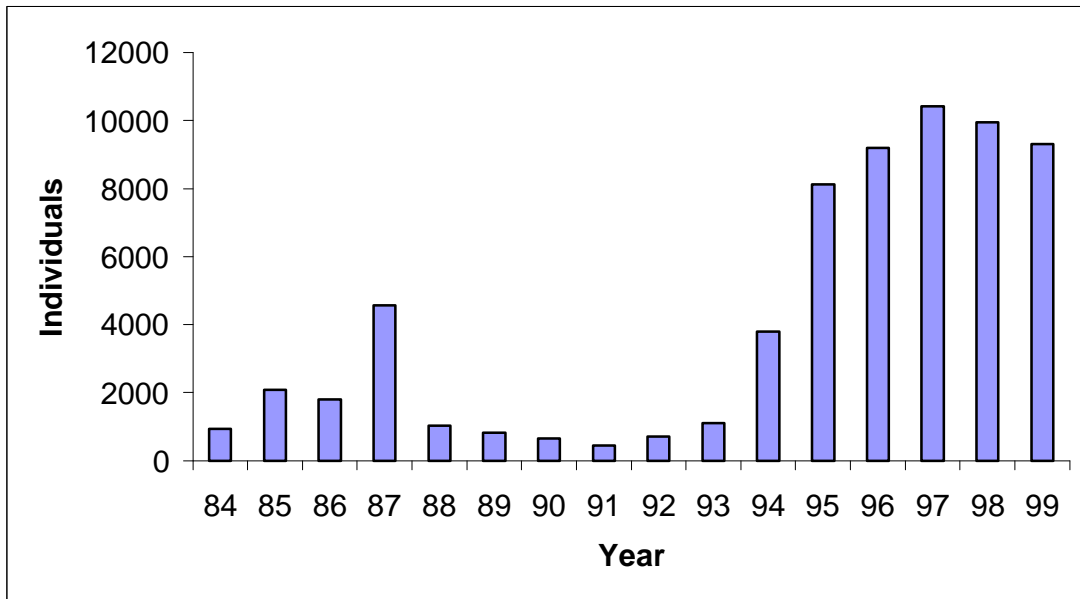


Figure 14. December estimates of DCCOs at the Salton Sea based on CBCs, 1984 – 1999.



Mexico

Summary of Population Data and Trends

(No survey completed for Mexico; population data from Carter et al. 1995)

Breeding.

The DCCO has been documented as a nesting species in Mexico since at least the beginning of the 20th century (see Historical Information, below). The southern limit of the known breeding range for *P.a. albociliatus* is at Bird Island, Sinaloa, but some birds may nest farther south (Carter et al. 1995). This subspecies is currently known to nest in Baja California (Norte and Sur), Sonora and Sinaloa, but no data were available post 1992. Birds nesting farther inland in adjoining Durango probably belong to populations in eastern North America (Carter et al. 1995). The following data were summarized by Carter et al. (1995).

In Baja California, 3,394 pairs bred at 20 colonies between 1968 – 1992, comprising 14 % of the Pacific Coast marine population. Most of these birds nested on the west coast of the Baja California peninsula. Some areas in this region have not yet been completely surveyed, especially Piedra near Guerro Negro and the Islas San Benito. Formerly, the largest known DCCO colony ever documented occurred on this peninsula, located at Isla San Martin, but was abandoned in the late 1970s (see Historical Information, below). Thus, this colony was not included in Carter et al.'s (1995) Pacific Coast total. The large colony at Isla Santa Margarita (Las Tijeras Mangrove) was one of the largest (currently active) on the Pacific Coast at the time of last count (1992).

In Sonora and Sinaloa, 3,575 pairs bred at seven known colonies from 1973 – 1991, and comprised 15 % of the Pacific Coast population. Large colonies at Isla Alcatraz, Sonora, and North Altamura, Sinaloa, were among the four largest currently active colonies on the west coast based on their totals (1500 pairs each) when they were last surveyed in 1975. At least 300 pairs were estimated to nest inland in Sonora at Rodriguez and El Molinito Reservoirs. Russell and Monson (1998) reported that the principal nesting in inland Sonora occurs along the Rio Yaaqui, wherever there are dead trees or brush to support nests.

Precise information is not available on population trends. However, in 1980, breeding numbers in the Gulf of California were roughly estimated at 10,000 pairs (Anderson 1983). The more recent estimates by Carter et al. (1995) are in that order of magnitude, but of more detail. Exact DCCO status in this region is basically unknown, but probably stable. Factors that might threaten these birds include: aquaculture operations which are increasing in DCCO habitat along the west coast of Mexico (threaten because cormorants would be expected to be controlled as they depredate aquaculture; the opposite might just as well be true, that DCCO are increasing due to this); the other effect of aquaculture is habitat destruction, where nesting sites have been disturbed or destroyed by increasing human activities in the mangrove and other swamps of western Mexico. It is also possible that DCCO have declined on some of the islands in the northern Gulf by disturbances and possibly even Newcastle disease, where it is likely in DCCOs in the northern Gulf (D.Anderson, pers. comm.).

Winter.

Howell and Webb (1995) reported that the DCCO is a fairly common year round resident along both coasts of Baja Sur. It occurs locally on the coast of northern Sonora, and is rare in winter south to Nayarit. In Durango it is an uncommon and local resident. From northwest Yucatan to northern Belize it is a fairly common resident. Along the Gulf Coast of Tamaulipas

and Campeche, it is a fairly common winter visitor, and is uncommon to rare to Veracruz and Tabasco. Occurs locally inland in northern Mexico to Sonora, Nuevo Leon and San Luis Potosi. No CBCs are conducted in Mexico.

Migration.

No information obtained.

Historical Information

Early in the 20th century, colonies were reported on just about every island on the west coast of Baja California (Grinnell 1928). The earliest nesting record we obtained is from Wright (1913) who estimated nearly 350,000 DCCO nests at San Martin Island in Baja California, the largest colony ever reported in North America. Wright (1913) reports seeing

“a steady stream of cormorants, flying about eight or ten abreast. This stream poured from these hills continuously and reached as far as we could see, toward the bay of San Quentin. The stream was like a great black ribbon that waved in the breeze and reached to the horizon...the birds kept coming as though there were no limit to their numbers...the flow of birds was continuous during the daylight hours of each day we were there. The flow was unbroken—simply one steady stream going, all day, and a steady stream returning.”

While Carter et al. (1995) state that Wright’s estimate was likely an overestimate due to incorrect extrapolation, it is clear that this colony, even if it was only 1/10 the size of Wright’s estimate, was enormous compared to currently existing DCCO colonies, or other cormorant colonies anywhere in the world. In the 1970s the colony still existed, but numbers were greatly reduced; counts obtained in 1969 and 1975 estimated 2,500 and 10,000 -12,000 breeding pairs, respectively. However, in 1977 there were no nests but many people were observed on the island. This historic colony is thought to have disappeared due to human disturbance, the introduction of alien predators, and to a smaller degree, contaminants (Carter et al. 1995).

Other early colonies were also reported to be fairly large. At Islas Los Coronados, about 1,000 pairs bred in the early 1900s, but by the mid 1920s only a few pairs remained due to human disturbance from tourism. At last count in 1991, 174 nests were counted at this site.

Overall, continued human disturbance, predations by feral cats, and persistent contaminants have resulted in low breeding success, colony abandonments, and shifts in nesting locations between years (Carter et al. 1995). While greater effort is needed to monitor Mexico’s nesting population, it is clear that it has been much reduced since the early 20th century.

Land Ownership

No information available.

Productivity

No information obtained.



Nevada

Summary of Population Data and Trends

(Survey completed by L. Neel, Nevada Division of Wildlife; numbers for Anaho Island colony provided by D. Withers, Stillwater NWR.)

Breeding.

The DCCO has been documented as a breeding species in Nevada since at least 1950 (Alcorn 1988). Breeding colonies occur in the southern and western portions of the state; since 1986 a total of five known colonies has been reported. The largest and most persistent of these is at Anaho Island NWR in Pyramid Lake; the number of nests reported between 1950 – 1998 ranged from a low of 200 to a high of 2500. Between 1986 – 1998, the average number of nests was 957; however, numbers appear to have declined since 1986 (Figure 15). Because regular statewide surveys are not conducted, overall population trends are not available.

Winter.

In winter, the DCCO occurs in southern Nevada, and was recorded as a permanent resident on the lakes in the Pahranaagat Valley and on the Colorado River in the 1950s (Alcorn 1988). No information was provided on current numbers or locations. Christmas Bird Counts conducted between 1984 – 1998 (Sauer et al. 1996) recorded only one site, Henderson, with counts of ≥ 100 birds in at least one year. During this time, numbers fluctuated between 15 – 110 birds. With the exception of 110 birds counted in 1996, numbers were < 100 birds in all years during this period.

Migration.

During migration DCCOs are transient over most of the state (Alcorn 1988). Significant concentrations are reported at Walker Lake (500 – 1000 in fall, 100 – 350 in spring) and Pyramid Lake (2,300 in fall).

Historical Information

No information obtained.

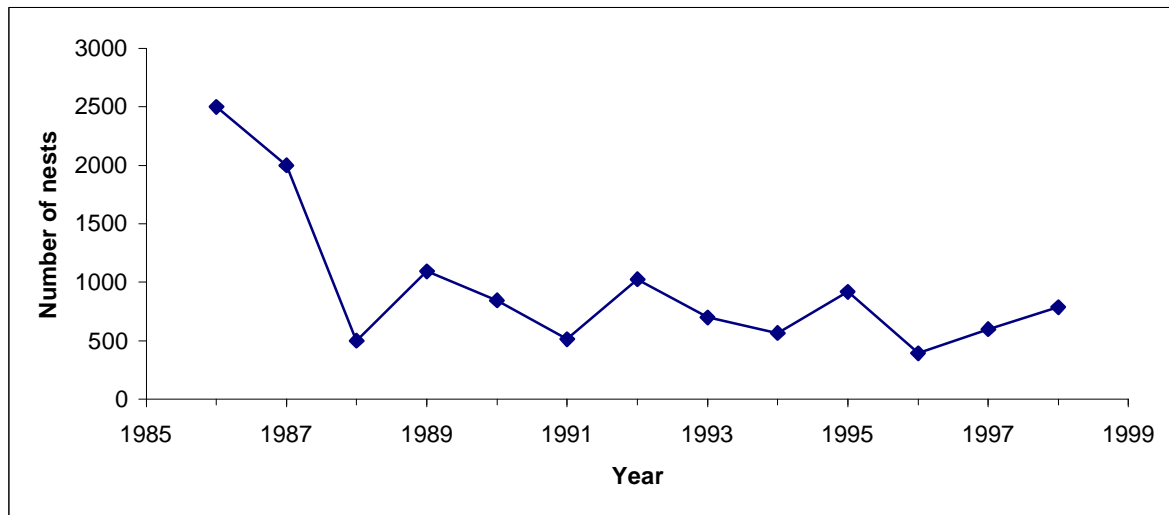
Land Ownership

Information provided for one colony at Humboldt Wildlife Management Area (WMA), which is federal land being leased by the state.

Productivity

No information provided.

Figure 15. Changes in number of breeding pairs at Anaho Island, Pyramid Lake, Nevada, 1986 – 1998.



New Mexico

Summary of Population Data and Trends

(Survey completed by S. Williams, New Mexico Game and Fish.)

Breeding.

The DCCO has been documented as a nesting species in New Mexico since 1937, but possibly occurred earlier (see Historical Information, below). Both *P.a. albociliatus* and *auritus* probably occur in New Mexico, a fact that may provide important information for delineation of breeding populations. In the Lower Rio Grande drainage birds tend to have the white nuptial plumes of *albociliatus*, while birds in the Upper Rio Grande drainage (Abiquiu Reservoir) and possibly elsewhere in the north tend to have the black nuptial plumes of *auritus* (S. Williams, pers. comm.). We included breeding birds occurring in the Lower Rio Grande drainage area (at least 82 % of breeders), in Zone 2; birds breeding in the Upper Rio Grande and Middle Pecos River were included in the summary of Zone 3. Data are still being gathered and analyzed to determine more specific information on subspecies delineation (S. Williams, pers. comm.).

The last statewide estimate for numbers of breeding pairs and colonies was made in 1996. At that time 730 pairs were estimated at about five major colony sites. DCCOs breed in three regions, all on reservoirs: 1) the Lower Rio Grande Valley, primarily at Elephant Butte and Caballo reservoirs and vicinity, about 600 pairs (there are multiple sites in this area, and birds shift locations from year to year); 2) the Upper Rio Grande, at Jemez and Abiquiu reservoirs,

about 100 pairs; and 3) the Middle Pecos Valley at Santa Rosa reservoir, about 30 pairs. In 1992 the same numbers of birds were estimated at the same colonies.

Precise information on population trends is not available due to scarcity of early data and lack of systematic nest searches. However, available data suggest the following trends: from the late 1930s to the mid 1970s no real change occurred in the Elephant Butte Reservoir area. Establishment of nesting sites in the Caballo Reservoir in the late 1970s combined with generally larger numbers noted in the Elephant Butte – Caballo region by the mid-1980s suggests that the nesting population may have increased in the lower Rio Grande Valley during that time. Two new colonies in the Upper Rio Grande Valley and one new colony in the Middle Pecos River Valley in the early to mid-1980s also suggests an increasing trend in New Mexico during that period. More recent reports suggest breeding numbers at Elephant / Caballo reservoirs have decreased in recent years, as snags fell and were not replaced. In the Upper Rio Grande Valley, Jemez Reservoir is scheduled to be drained and abandoned, so this colony may disappear (S. Williams, pers. comm.).

Winter.

Significant winter concentrations of DCCOs occur in the Elephant Butte / Caballo (1000+ birds) and Brantley – Avalon (1000+ birds) areas (Carlsbad areas). The maximum number of birds estimated to winter in the state is 2000+ individuals, about equally distributed in the two areas. However, winter numbers vary from year to year, depending on water levels, food availability, and other factors (S. Williams, pers. comm.). CBCs conducted between 1984 – 1998 recorded three sites where ≥ 100 DCCOs were counted in at least one year (the two sites mentioned above and at Loving). At the latter site, an unusually high count of 805 birds was recorded in 1996; in most years counts at this site total < 10 birds. Large annual variation in winter numbers is apparent in the CBCs (Figure 16), and no trend information is available. In recent years, numbers in the Elephant / Caballo area may be lower, while in the Brantley – Avalon area numbers may be higher (S. Williams, pers. comm.). Data from CBCs at these sites conducted between 1996 – 1998 support this statement.

Migration.

Information on migrants in New Mexico is very limited. Migrants may occur, but extent of occurrence and number of migrants is not well understood (S. Williams, pers. comm.).

Historical Information

The DCCO was first recorded in the state in 1913 when one of three individuals was collected in December near present day Truth or Consequences, Sierra County. First breeding record was obtained in 1937 at Elephant Butte Reservoir area, when two colonies totaling 250 “nesting” were reported. (It is not clear if number reported referred to individuals or pairs.) However, DCCOs were possibly breeding in the same vicinity in 1932 (S. Williams, pers. comm.). From the late 1930s on it was presumed to be resident in this area. Between 1950 – 1963 up to 24 pairs nested at Bosque del Apache NWR in Socorro County, but this area was abandoned due to habitat changes. In 1975, 260 nests were estimated at Elephant Butte Marsh, and in 1979, 300 nests were estimated at Caballo Reservoir. In 1985 surveys estimated about 450 nests at these two reservoirs. Colonies in the Upper Rio Grande Drainage were discovered in the 1980s and have been regular since then. The colony at Santa Rosa Reservoir in the Middle Pecos River drainage was discovered in 1985 and has also been regular.

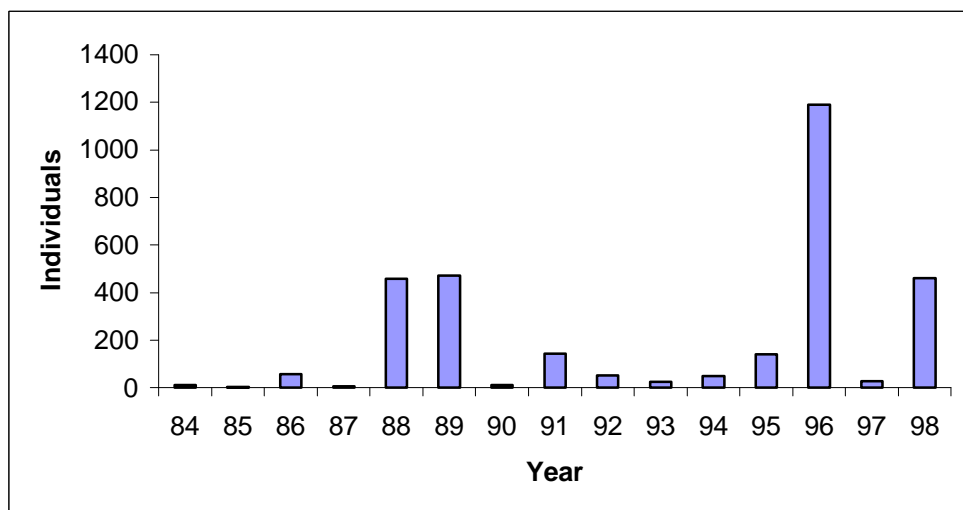
Land Ownership

Reservoirs where breeding colonies occur are assumed to be on public lands, managed by the U.S. Bureau of Reclamation and the U.S. Army Corps of Engineers (USACE) (S. Williams., pers. comm.).

Productivity

No information provided.

Figure 16. December estimates of DCCOs in New Mexico based on CBCs, 1984 – 1998.



Oregon

Summary of Population Data and Trends

(Survey completed by R. Lowe, USFWS; most data on breeding birds from R. Lowe, USFWS, M. Naughton, USFWS, and Carter et al. (1995); additional data supplied by M. Lawes, USFWS, J. Hainline, USFWS, and A. Clark, USFWS.)

Breeding.

The DCCO has been documented as a nesting species in Oregon since at least the late 1800s (see Historical Information, below). Between 1979 and 2000, approximately 40 known colony sites were active. While annual statewide censuses for breeding DCCOs are not regularly conducted, Carter et al. (1995) summarized trends for the breeding regions in Oregon through 1992, and these data are summarized below.

Carter et al. (1995) divided the coast into four regions: the Columbia River estuary, the north coast, the central coast, and the southern coast. Between 1988 – 1992 an estimated 6,249 pairs nested along this coastal region, comprising 26 % of the Pacific Coast marine population.

The largest numbers occur in the Columbia River estuary, where large increases have occurred in the last 20 years. Though DCCOs were not documented nesting in the estuary until 1980, by 1992, more than half of Oregon's DCCOs nested here. Up until 1998, the greatest concentrations have occurred in two colonies in the estuary: East Sand Island, the largest (currently active) colony on the Pacific Coast, and Rice Island. In 1998, however, numbers declined on Rice Island and increased on East Sand Island to 6147 breeding pairs (Collis et al. 1999). In 1999, no DCCOs nested on Rice Island, and numbers on East Sand increased to an estimated 7,242 pairs (D. Roby and K. Collis, pers. comm.). In this year management was initiated to discourage Caspian Terns from nesting on Rice Island, and DCCOs from Rice Island are thought to have immigrated to East Sand Island, probably as a result of these efforts. In 2000, East Sand Island supported an estimated 6390 breeding pairs (D. Roby pers. comm.). Double-crested Cormorant numbers in the Estuary declined for unknown reasons (Figure 17), and again no nesting occurred on Rice Island (D. Roby, pers. comm.).

Along other regions of the coast increases also occurred. On the north coast, 983 pairs were estimated between 1988 – 1992, primarily at four large colonies on islands and mainland cliffs. For this region, no data were provided post-1992. On the central coast, 599 pairs were estimated at four active colonies on islands and the mainland; increases occurred at three of the four colonies. With the exception of a small new colony (4 pairs) found at Yaquina Bay in 2000, no data are available for this area post-1992. On the southern coast, 1,357 pairs were estimated at 11 active colonies; increases occurred at the majority of these colonies. With the exception of two of these colonies, plus a new colony found at the McCullough Tower, no data are available for this area post-1992..

In the interior, the most significant interior colonies are at Malheur and Upper Klamath Lakes; between 1986 – 1999 numbers at both of these colonies underwent significant declines (Figure 18). In 1986, the combined total of breeding pairs at these lakes was 1,882, while in 1999 the combined total was 759 pairs. The interior lake region of eastern Oregon is a very dynamic ecosystem and the availability of nesting habitat near water fluctuates in response to wet and dry climate cycles. For example, Malheur Lake increased from 40,000 acres to > 175,000 acres between 1982 and 1986. Drought conditions since the late 1980s caused the lake to decrease in size to about 300 acres in 1992 (Carter et al. 1995). DCCO populations respond to these changes in habitat and consequently breeding populations fluctuate (M. Naughton, pers. comm.). The only other interior colony that appeared to be censused regularly was at Summer Lake, where the highest number of pairs estimated during counts between 1988 – 2000 was 60 pairs.

Winter.

In winter DCCOs are reported to occur over much of their breeding range, except where water freezes over (Gabrielson and Jewett 1940). Along the coast DCCOs commonly occur in virtually every estuary, some open ocean locations and most freshwater lakes. Coastal locations that may support 500 or more wintering DCCOs include the lower Columbia River, Coos Bay, and Siltcoos Lake (R. Lowe, pers. comm.). No information is available on maximum numbers, and no current information was provided or obtained on wintering birds in the interior. Other coastal estuaries (e.g., Nestucca, Tillamook) could potentially support larger numbers of DCCOs and other cormorant species, however, a state supported hazing program operates to harass cormorants away from these areas (M. Naughton, pers. comm.). CBCs conducted between 1959 – 1988 suggest an increase of 1.9 % per year (Sauer et al. 1996). Counts conducted between 1984 – 1998 recorded DCCOs at 14 sites, with a high of 2,746 birds for the state in 1993 (Figure 19). During these 15 counts large numbers (> 500 birds) were recorded at Sauvie Island and in

the Columbia Estuary. These more recent data have not been analyzed for trends by the Patuxent Wildlife Research Center, and no trends are readily apparent from Figure 19.

Migration.

During migration, flocks moving along the Oregon coast are routinely observed in spring and fall. However, information on the number of migrants and peak periods is not available. While some DCCOs in this area are undoubtedly year-round residents, it is believed that significant numbers do migrate (R. Lowe, pers. comm.). Limited data from chicks banded at East Sand Island suggest that birds migrate to Puget Sound and also south of their natal colony (A. Clark, pers. comm.).

Historical Information

The first definite record of the DCCO in the Columbia River is from Townsend (1839), though Lewis and Clark (1814) reported “cormorants” at the mouth of the Columbia River as early as 1805. Early breeding records from 1875 – 1888 describe “huge” colonies in Oregon’s “lake counties” (Gabrielson and Jewett 1940). On the north coast, Finley (1902, 1905) reported DCCOs as the second most abundant cormorant nesting at middle and west rocks of Three Arch Rocks around 1900, but nesting in this area has greatly declined (Carter et al. 1995). Declines also occurred at Haystack Rock between 1930 – 1957, and remained low until 1988, after which time numbers began to increase again. In the Columbia River Estuary the dramatic increases since 1980 probably reflected movement from interior nesting areas, and some birds may have immigrated from Grays Harbor, Washington. In the interior portion of the state at Malheur Lake, breeding numbers averaged 200 pairs in the early 1980s, but increased dramatically to 1120 pairs by 1987. This increase was linked to creation of new feeding and nesting habitats from 1982 – 1986. However, severe drought conditions since the late 1980s caused the lake area to drop and in 1992, only 50 pairs attempted to nest. Changes in DCCO numbers at this lake are thought to be representative of changes at other lake colonies in Oregon, California and Utah, while increases on the outer coast of the state may be related to immigration from interior populations (both within and outside Oregon). Other factors that contributed to recent increases in DCCO numbers may include reduced human persecution and protection of nesting habitat, though some fishermen have continued to harass cormorant species in Oregon (Carter et al. 1992).

Land Ownership

Ownership of coastal colonies is primarily public, however, seven colonies are on private land. The breakdown of public ownership: USFWS/NWR (19), USACE (1), US Coast Guard (1), State of Oregon (4), unknown public (3). The three unknown sites are Miller Sands Navigational Aids, Trestle Bay and Desdemona Sands Pilings. East sand Island is owned by USACE; Rice Island is owned by the State of Oregon (M. Naughton, pers. comm.). The three wintering sites are all managed by the State of Oregon.

Productivity

In 1997, Double-crested Cormorant nests located on channel markers in the vicinity of Rice Island (n = 64) fledged an average of 1.55 young per initiated nest. Active nests on East Sand Island (n = 50) that were checked late in the chick-rearing period contained an average of 1.61 young, while the comparable number on the successful channel marker nests near Rice Island was 2.11 young (Roby et al. 1998). Cormorant nesting productivity within the estuary in 1998 was sampled by monitoring nests (n = 70) in each of three different locations: the channel

markers in the vicinity of Rice Island, on Rice Island and on East Sand Island; nests in each location fledged an average of 1.59, 0.55, and 1.23 young per initiated nest, respectively (Collis et al. 1999). In 2000, the channel markers in the vicinity of Rice Island experienced a sharp decline in productivity (0.43 young per initiated nest; n = 75; Anderson et al., unpubl. data). The East Sand Island cormorant colony fledged an average of 1.20 young per initiated nest during 2000 (n = 40) (C. Anderson pers. comm.). This was similar reproductive success as on East Sand Island in 1998, although it was significantly lower than the channel marker nests in 1997 and 1998 and significantly higher than on Rice Island in 1998.

Figure 17. Changes in number of breeding pairs at East Sand and Rice Islands, Columbia River Estuary, Oregon, 1991 – 2000.

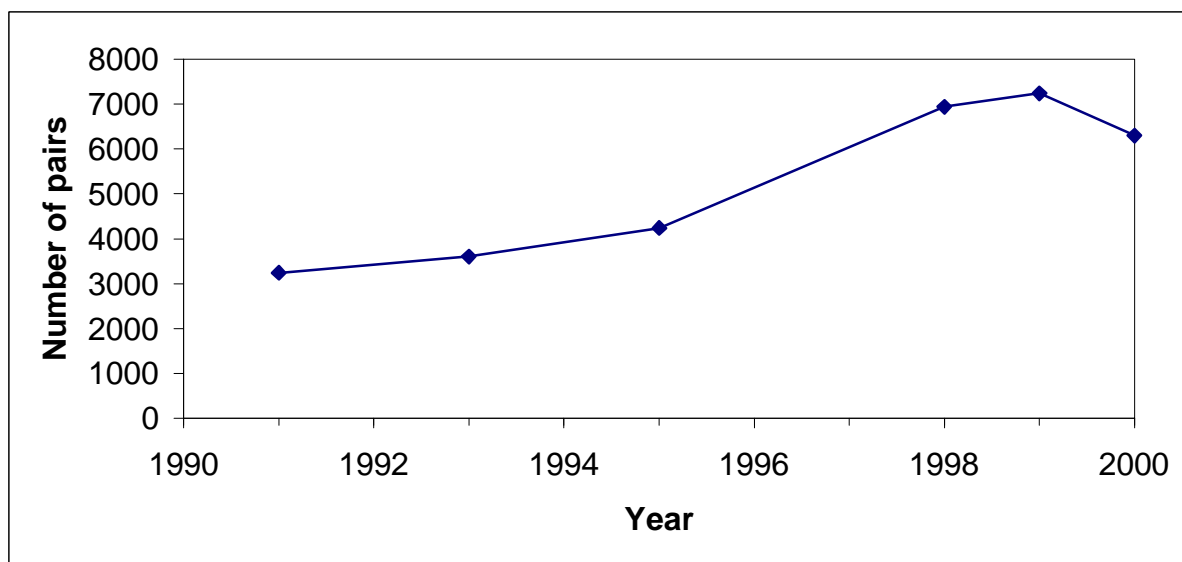


Figure 18. Changes in numbers of breeding pairs at Malheur and Upper Klamath Lakes, 1986 – 1999.

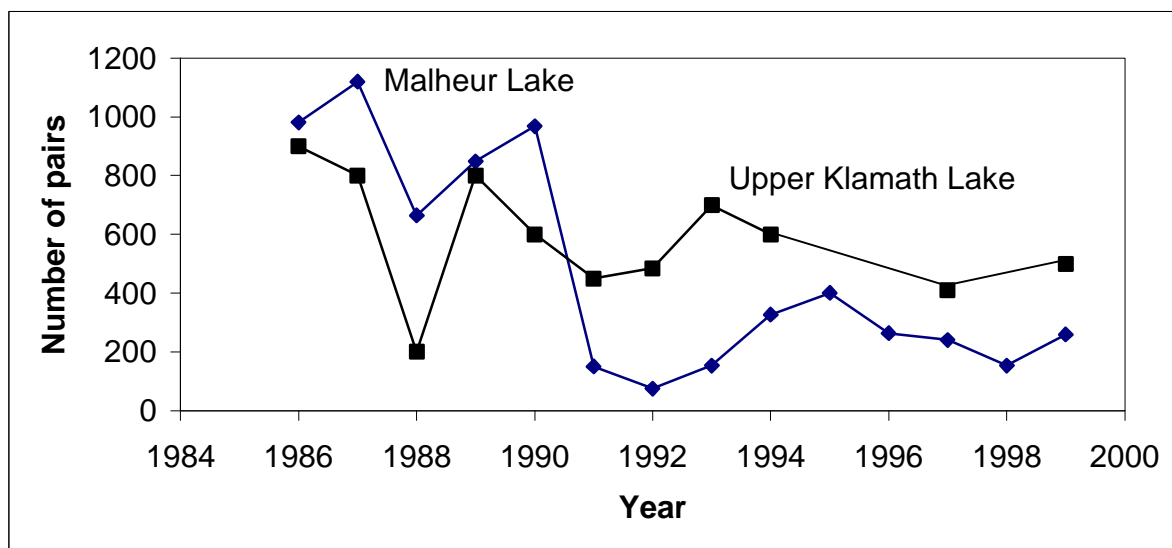
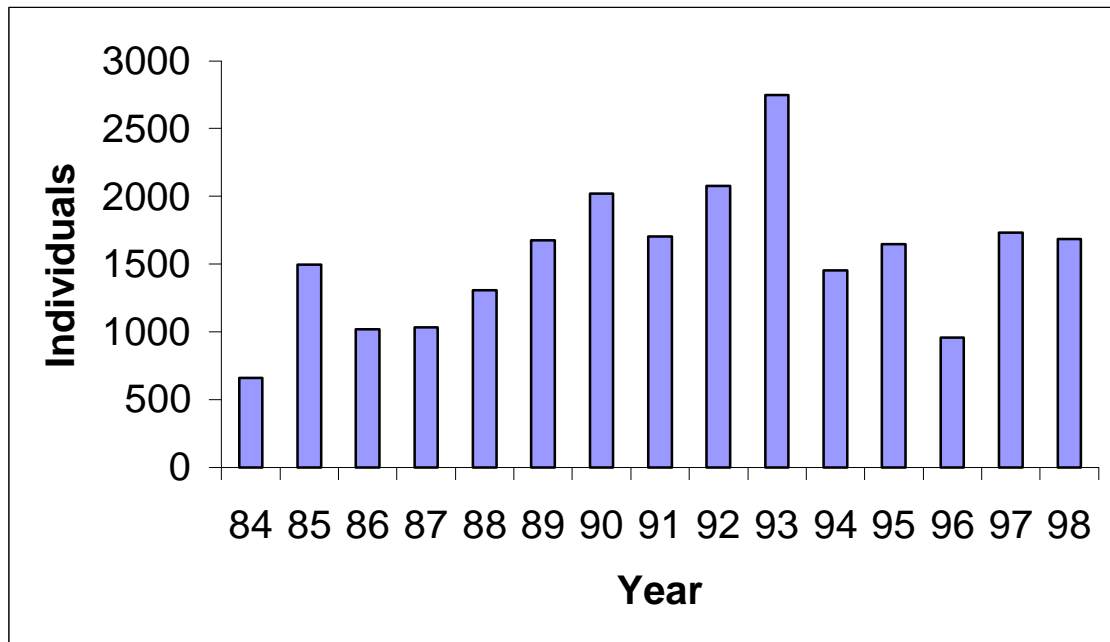


Figure 19. December estimates of DCCOs in Oregon based on CBCs, 1984 – 1998.



Washington

Summary of Population Data and Trends

(No survey completed; population data from U. Wilson (USFWS) and summarized from Carter et al. 1995.).

Breeding.

The DCCO has been documented as a nesting species in Washington since at least 1907 (see Historical Information, below). Since 1970, a total of 43 breeding colonies has been documented. Nearly all are on offshore islands, and nests are mainly on the ground or on cliffs. Carter et al. (1995) reported results of surveys conducted in 1992 for five breeding regions along the coast: the San Juan Islands, Juan de Fuca Strait East, Olympic Peninsula Outer coast, Grays Harbor Bay, and the Columbia River mouth. An estimated 1,618 pairs nested at 22 colonies, and comprised about 7 % of the Pacific Coast marine population. Fairly large colonies (50 – 350 pairs) were observed in the San Juan Islands, the Juan de Fuca Strait East, on the Olympic Peninsula, and in the Grays Harbor Estuary at Goose and Unnamed Sand Islands; about 54 pairs were estimated in the Columbia River mouth. The largest numbers were reported for Juan de Fuca Strait East (33 %) and the Olympic Peninsula Outer Coast (35 %). Between 1975 – 1992, numbers of breeding DCCOs increased on both the inner and outer coastal waters, though increases were more pronounced on inner waters. Carter et al. (1995) note that Wilson (1991)

showed few birds nested on the outer coast during strong El Nino years in the 1980s, and numbers there declined during and after El Nino years. Carter et al. (1995) suggest that birds may move to inner waters during El Nino years, and back to outer coast waters afterwards. Therefore, El Nino conditions that occurred off the Pacific Coast in 1992 may have affected the distribution and numbers of breeding DCCOs recorded for different parts of the Washington coast (Carter et al. 1995).

Although an increasing trend was observed for the Washington coastal colonies from 1975 - 1992 (Carter et al. 1995), overall there was no significant trend from 1979 - 2000 (U. Wilson, pers. comm.). Along the Washington outer coast of the Olympic Peninsula, DCCO breeding populations peaked in 1990 (Figure 20). Between 1990 - 2000 a significant decline of about 72% occurred in this area ($r_s = -0.718$, $n = 11$, $p < 0.02$) (U. Wilson, unpubl. data). More inland, in the eastern part of Juan de Fuca Strait and the San Juan Islands, colonies showed no trends from 1983 - 2000 ($r_s = -0.110$, $n = 18$, $p > 0.5$) (Figure 21, U. Wilson, unpubl. data). Along the south central coast, the Grays Harbor colonies have almost disappeared since 1992 when 440 pairs nested here (Carter et al. 1995). Goose Island, a former breeding site, has entirely washed away. Sand Island supports few nesting DCCOs; approximately 10 pairs nested in 2000 (D. Roby, pers. comm.). Some birds continue to nest on the pilings and navigational aids in Grays Harbor. DCCOs in Washington frequently shift colony locations. The Lower Columbia River colonies (East Sand and Rice Island) increased in size significantly during the period 1990 - 2000. These colonies are located in Oregon, at the Oregon/Washington border, and it is likely that birds moved from Washington to the Lower Columbia River colonies (M. Naughton, pers. comm.).

In the interior, numbers were available for only one colony, located at the Northern Potholes reservoir. This colony was last counted in 1997, at which time 652 pairs were estimated (J. Stofel and J. Tabor, pers. comm.). At this site increases have occurred steadily since 1978 (Figure 22).

Winter.

In winter, thousands of DCCOs occur on Puget Sound. Two series of surveys in this area found a 62% decrease in numbers between 1978/1980 and 1992/1999 (D. Nysewander, pers. comm.). The species also occurs east to Seattle, south on Puget Sound to Tacoma and Steilacoom, and on the open coast to Cape Disappointment (Jewett et al. 1953). Larrison and Sonnenberg (1968) describe it as a fall, winter and spring visitor to Puget Sound and to inland lakes (Sept. to late May). CBCs conducted between 1959 - 1988 suggest an increase of 10.4 % per year (Sauer et al. 1996). More recent counts, conducted between 1984 - 1998, have not been analyzed for trends by the Patuxent Wildlife Research Center, but suggest increases continue to occur; a high of 6,821 birds was recorded in 1995 (Figure 23). During these more recent counts, DCCOs have been recorded at 21 sites, with high numbers (> 500 birds) counted in Tacoma, Seattle, Kitsap County, Padilla Bay, San Juan Islands Archipelago and Bellingham.

Migration.

No information obtained.

Historical Information

Carter et al. (1995) reported Historical Information for DCCOs in Washington, and this information is summarized below. The earliest nesting records we obtained were collected between 1907 - 1915, when 12 colonies (about 400 pairs) on the outer coast of Washington were

reported. Nesting was documented at White Rock and Bird Rocks when breeding expanded to the San Juan Islands in 1937. Between the 1940s and 1970s, 12 new colonies appeared in this area. But beginning in the 1960s and 1970s, many of these colonies were abandoned; by 1980 only four colonies were still active. Between 1983 – 1986, three disappeared and only Bird Rocks remained active. Increasing human disturbance since the 1950s is thought to be the cause of these declines. In the Juan de Fuca Strait colonies on Smith and Protection Islands have experienced severe human disturbance and nest predation by immature Bald Eagles, which resulted in no productivity from 1990 – 1992. Despite these events, numbers increased in these areas up to 1992. Between 1997 – 1999, numbers at Smith and Protection Islands underwent significant declines (see above). On the outer coast, numbers reported in the late 1970s were similar to those reported for the 1907 – 1915 period, though colony location changed.

In eastern Washington, DCCOs have been present since at least 1932 in the south-central Columbia Basin (Smith et al. 1997), but we did not determine status at that time.

In 1992, eggshell thinning was detected in eggs at Goose Island at the Grays Harbor estuary (Kiff 1994).

Land Ownership

No information obtained or provided.

Productivity

Henny et al. (1989) made observations on nest success and productivity on Colville Island in the San Juan Islands in 1984. They did not follow individual nests, but recorded that 66 young from 230 nests (0.29 young / occupied nest) were reared to two-thirds of adult size.

Figure 20. Changes in breeding numbers at colonies located along the Washington outer coast, 1979-1999.

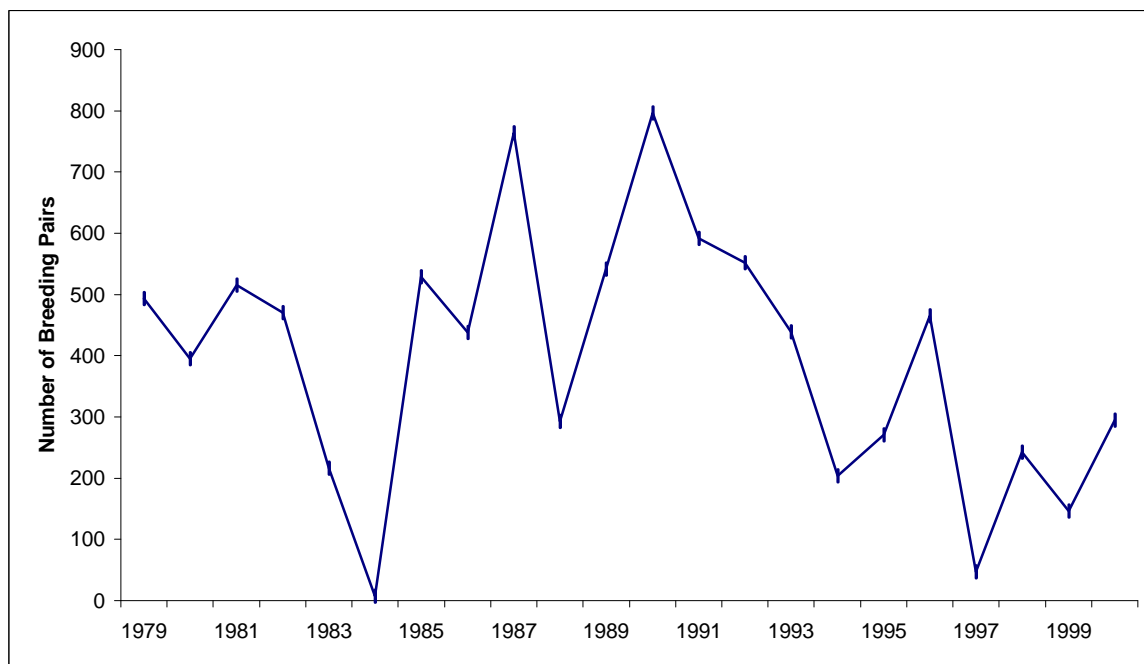


Figure 21. Changes in breeding numbers at colonies located in the Washington inner coastal waters, 1983-2000

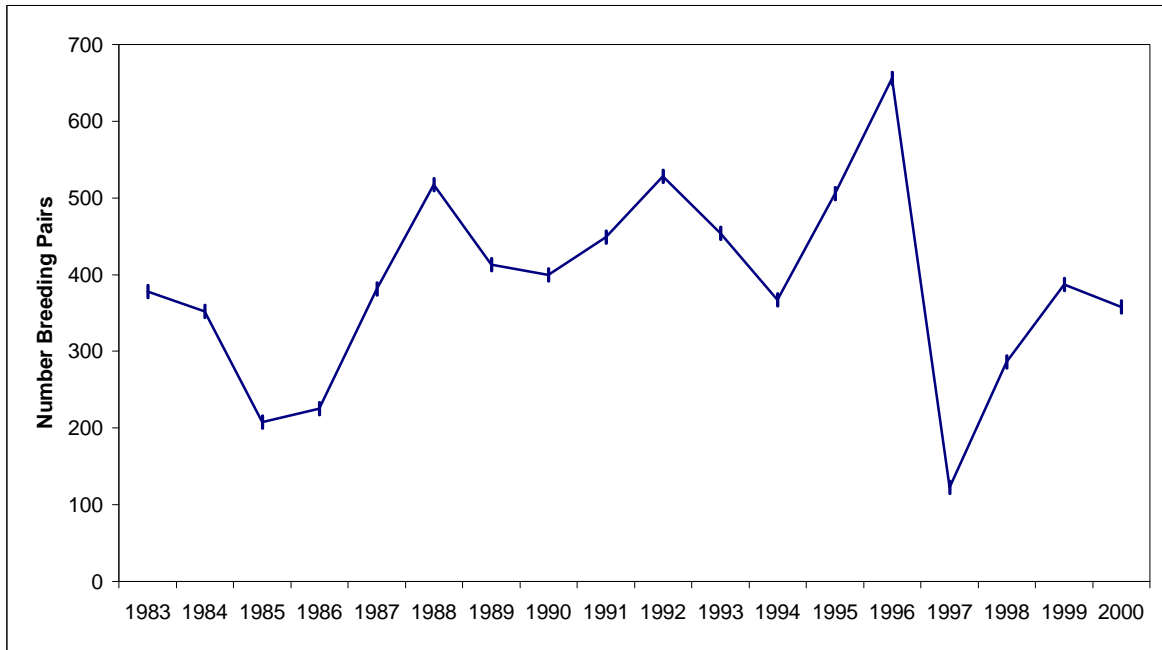


Figure 22. Changes in breeding numbers in interior Washington, N. Potholes Reservoir, 1978 – 1997.

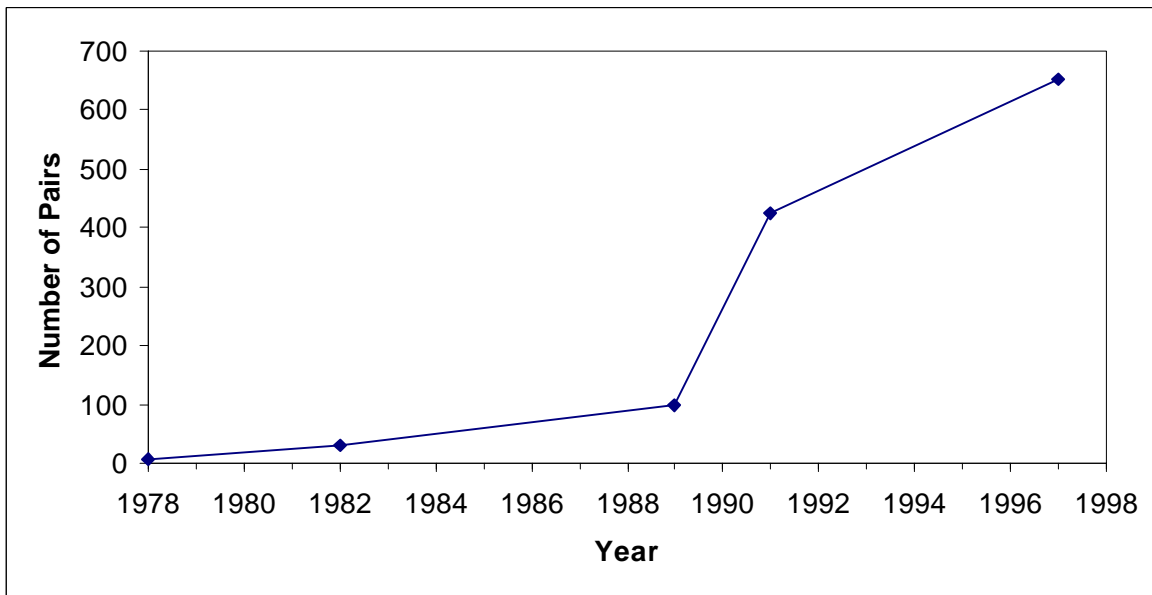
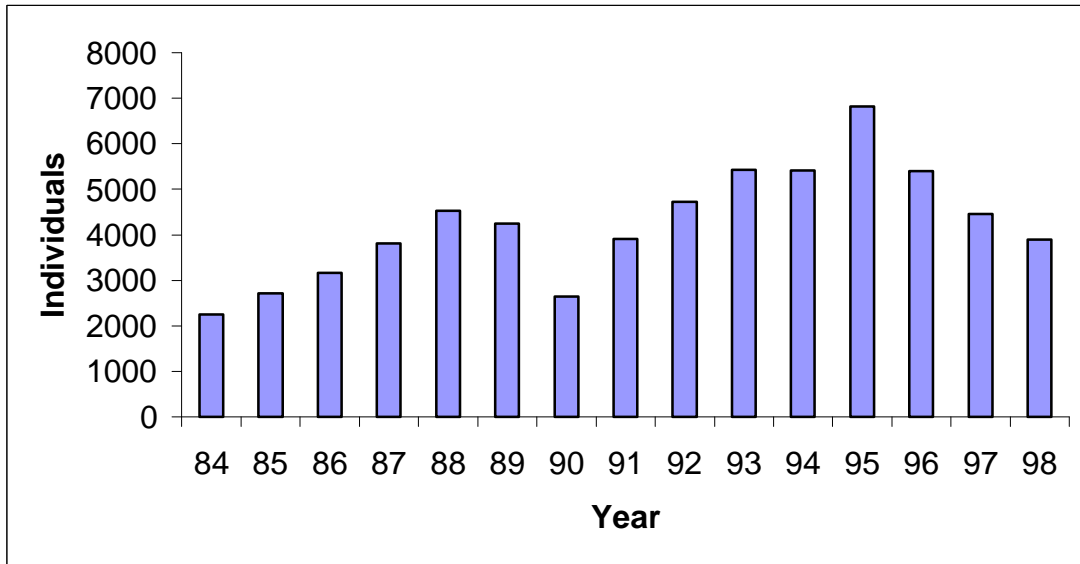


Figure 23. December estimates of DCCOs in Washington based on CBCs, 1984 – 1998.



ZONE 3: SUMMARY OF POPULATION DATA FOR CANADIAN AND U.S. INTERIOR POPULATION

Introduction

Breeding Information

In Zone 3, the breeding population spans across the Prairie Provinces of Canada (Alberta, Saskatchewan and Manitoba), the Canadian and U.S. Great Lakes and southwestern Québec, and extends west of Minnesota to sw. Idaho; range extends as far south as central Utah, and central Colorado. Also breeds locally in central Kansas and possibly northern New Mexico. Table 4 shows the status (Breeding, Wintering, Migrant) of the Double-crested Cormorant in this zone. Some of the birds included in this region, those from Idaho and Utah, show characters intermediate between *P.a. albociliatus* and *auritus*. In Idaho, 11 specimens were examined, and found to be intermediate in their characters between these two races, but closer to *auritus* (Burleigh 1972). In Utah, Behle (1941) reached a similar conclusion for the birds of Great Salt Lake. Thus we included these birds as part of Zone 3. However, in these areas some mixing obviously occurs between races. In New Mexico, the majority of birds breed in the Lower Rio Grande drainage and are believed to be *P.a. albociliatus*, while the small number that breed in the Upper Rio Grande drainage and possibly elsewhere in the north are thought to represent *P.a. auritus*. Because the majority of birds breeding in New Mexico are thought to be representative of *P.a. albociliatus*, New Mexico birds were grouped in the Pacific Coast population (see summaries for Zone 2: Summary of Population Data for the Pacific Coast and New Mexico).

Over the last two decades the number of breeding Double-crested Cormorants has greatly increased in the interior portion of the continent. Counts in the 1990s (mostly 1997) estimated a minimum of over 150,000 pairs in the region. This is a substantial underestimate, since numbers of breeding birds were not available for several areas, including most of Manitoba, where we know large numbers of DCCOs breed. Of 24 states and provinces for which we obtained data on breeding cormorants, 10 had increasing cormorant numbers, one had declining numbers, one had possibly stable or increasing numbers, and one had possibly stable / declining numbers. Data were not available for the other 11 locations to determine trends. Individual summaries for each state and province in this zone are included at the end of this summary and are in alphabetical order.

Prairie Provinces.

Very large numbers breed at Lake Winnipegosis, and Manitoba has the largest number of breeding cormorants in North America. However, complete province-wide surveys have not been regularly conducted in this province, so trends cannot be clearly demonstrated. Currently numbers appear to be fairly stable on Lake Winnipegosis in Manitoba, and are increasing in Saskatchewan and Alberta. Current numbers for the Prairie Provinces as a whole are not available; see provincial accounts for specific population data.

Great Lakes.

In the Great Lakes, increases have been dramatic since the 1980s (Figure 24), especially in Ontario, Michigan and Wisconsin waters, where the largest concentrations of breeding birds occur. Many of the Great Lakes states and Ontario have conducted regular state and province-wide surveys over several years. From 1970 to 1991, the population in the Great Lakes climbed

from 89 to 38,115 pairs, an average annual increase of 33 %. Growth in the Great Lakes has substantially declined in the last decade, even though numbers of cormorants are still increasing. From 1991 to 1997, the population went from 38,115 to 88,000 pairs, an average annual increase of 15 %. In 1997, about 75 % of these birds were in provincial Ontario and state of Michigan waters.

The most dramatic increases have occurred on Ontario, Michigan and Wisconsin waters. In some Ontario locations, growth appears to be slowing, which may indicate that numbers are approaching asymptotic values (Korfanty et al. 1997). While growth in Michigan and Wisconsin has also slowed from its very rapid initial increase during the first 10 years of the cormorant's recovery, numbers of cormorants rapidly increased in these states through 1997. Based on surveys of breeding colonies in 2000 for D. Trexel's M.S. thesis, growth rates on US waters of lakes Huron and Michigan between 1997 and 2000 appear to have declined even further.

Winter and Migrant Birds

No substantial numbers winter in this zone. We reviewed CBC data collected between 1984 – 1998 (Sauer et al. 1996) for all states that bordered Zones 2, 4 and 5 (Idaho, Utah, Colorado, Kansas, Missouri, Illinois, Indiana, and Ohio). Sites with ≥ 100 individuals during at least one count were identified in Idaho, Kansas, Missouri and Illinois, but there were not enough consistent data to create figures for these states. Overall, small numbers were occasionally observed during CBCs in the states reviewed, with a few exceptions (see state profiles). Substantial numbers are reported during migration, with as many as 25,000 – 50,000 birds reported in some of the Great Lakes states.

Historical Information

The earliest breeding record we have for this zone comes from Lake of the Woods, Ontario, where the species was first recorded breeding in 1798. Early breeding records (19th century) are also available for Manitoba and Saskatchewan, and for many of the Great Lakes states.

Great Lakes.

It is not exactly clear when the species began breeding on the Great Lakes themselves. Weseloh et al. (1995) note that breeding was not suspected to occur on the Lakes before 1913, when breeding was first documented on the far western end of Lake Superior. Colonies then spread slowly eastward. However, the second edition of the American Ornithologists' Union Checklist of North American Birds (1895) states the breeding range of the Double-crested Cormorant as "the Bay of Fundy, the Great Lakes, Minnesota and Dakota northward." While it is not clear what records were used to support the claim of breeding on the Great Lakes at this time, individuals involved on the AOU committee to revise matters relating to distribution included Elliot Coues, J.A. Allen, William Brewster, C. Hart Merriam, and Robert Ridgway, prominent and careful ornithologists. Additionally, breeding was well documented in many of the states surrounding the Great Lakes, and the species appears to have been abundant throughout the region during the 1800s (Lewis 1929). Therefore, it seems likely that the Double-crested Cormorant may have nested on the Great Lakes earlier than 1913, but may have been extirpated from the area before good documentation was obtained. A review of literature by early Great Lakes explorers may shed light on this question.

By the turn of the century, the Double-crested Cormorant had been greatly reduced or extirpated as a breeding species from the Great Lakes states (MN, OH, WI, IN, IL), Missouri,

Iowa and probably several of the prairie states. In some areas, migrants were also severely reduced. These declines appeared due to human persecution and exploitation. However, the few cormorants that remained in the region soon began a population expansion into or a re-colonization of the Great Lakes. Nesting was reported in Lake Superior between 1913 – 1920, reached lakes Ontario and Erie by the late 1930s, and the Upper St. Lawrence River in 1945 (Ludwig 1989; Weseloh et al. 1995). Numbers increased steadily during the 1930s and 1940s, and by the early 1950s cormorants had become so abundant in some areas of Ontario that control measures were authorized to reduce perceived competition with sport and commercial fisheries (Weseloh and Collier 1995). Aggressive illegal control measures also began around this time. Persecution from commercial fishermen and predation during nesting probably kept the numbers of birds breeding in Michigan and Canadian waters of Lake Huron at low levels (Ludwig and Summer 1995). Controls largely ended by around 1960, and probably only slowed the growth of the population in this region.

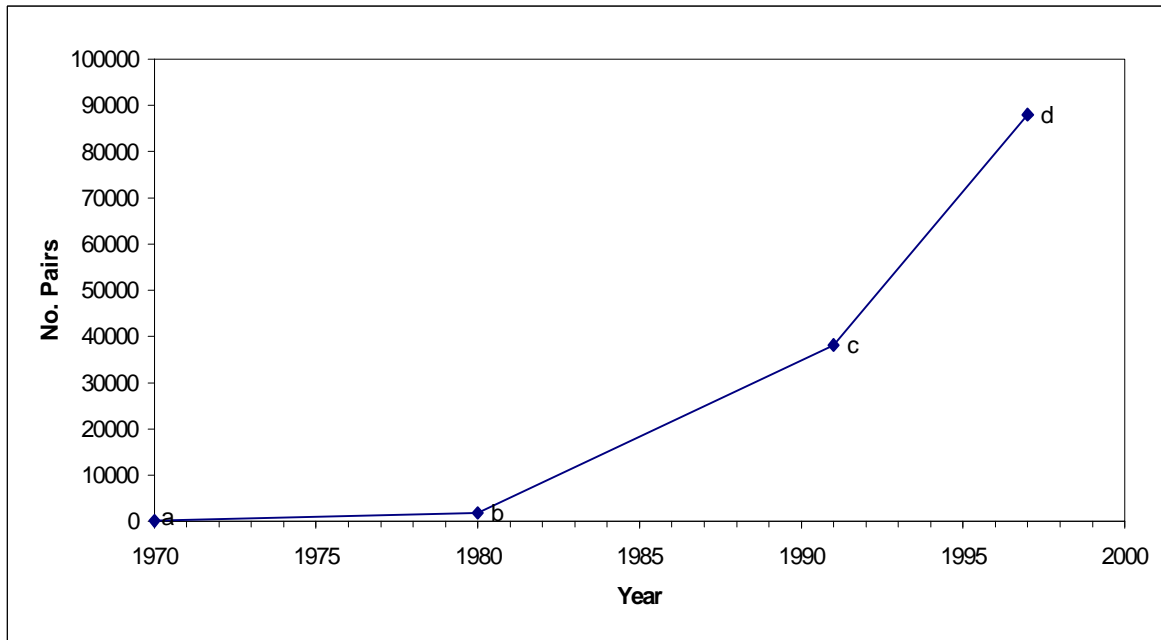
By 1960, even after control measures stopped, the cormorant population declined and continued to do so through the early 1970s. Cormorants disappeared from Lakes Michigan and Superior as a breeding species and declined rapidly in Lake Huron. In 1970, the population in the Great Lakes was estimated at only 89 pairs, nesting at eight sites (Weseloh et al. 1995). These dramatic declines are mostly attributed to high levels of toxic contaminants, such as DDE and PCBs, that were present in the Great Lakes and elsewhere throughout the annual range. These contaminants resulted in eggshell thinning and reproductive failure (Weseloh et al. 1983; Weseloh and Collier 1995; Ludwig and Summer 1995).

In the mid 1970s, cormorant numbers began to recover, due to a number of factors. Lower contaminant levels in the environment resulted from legislation implemented around this time to restrict the use of DDT and related pesticides (Weseloh and Collier 1995). Additionally, the DCCO was added to the list of birds protected by the Migratory Bird Treaty Act in 1972. Human induced changes in fish communities in the breeding range, development of aquaculture on the winter range, and creation of additional habitat also likely contributed to the recovery.

Prairie Provinces.

In the Prairie Provinces of Canada, certain colonies also appeared to decline around the turn of the century, but numbers did not drop as drastically as in the U.S. interior. In many areas cormorants remained relatively abundant. As in the Great Lakes, cormorant numbers reached high levels in the mid part of this century; from this point on cormorants have a history similar to that of the Great Lakes. In 1943 the Manitoba government employed cormorant control on Lake Winnipegosis, as the species was perceived to be responsible for losses to commercial fisheries. Numbers were greatly reduced by the early 1950s, when control stopped, but cormorant numbers continued to decline through the 1960s. Cormorants also declined in Saskatchewan and Alberta. Vermeer (1973) estimated a total of 6500 nesting birds in the prairie provinces for the 1967 – 1972 period, which is only about 2/3 the number counted on Lake Winnipegosis alone in 1945. Contaminants, illegal control and habitat loss were thought to be the main factors contributing to these declines (Vermeer and Rankin 1984; Koonz and Rakowski 1985; Korfanty et al. 1997). Cormorants received protection in all of these provinces during the 1970s and early 1980s, and numbers began increasing again. The greatest changes in numbers occurred in Saskatchewan and Manitoba, where populations increased from 5,850 to 32,558 pairs during the 1970s and early 1980s (Vermeer and Rankin 1984).

Figure 24. Changes in numbers of breeding pairs on the Great Lakes, 1970 – 1997.



a = 89 pairs (Weseloh et al. 1995)

b = 1798 pairs (Ludwig 1984)

c = 38,115 pairs (Weseloh et al. 1995)

d = 88,000 pairs (this report)

Table 4. Zone 3, Canadian and U.S. Interior. Summary of Population Data and Trends.

(+ = increasing; - = decreasing; 0 = no change).

State/ Province	Status (B, M, W, w) ¹	Number colony sites/ time period	No. active colonies/ last yr. surveyed	No. pairs/ last year surveyed	Breed. trend	Winter number (birds)	Migrant number (birds)
AB ²	B	NA	NA	NA	+	NA	NA
CO	B, M, w	21/1987-95	11/1992	1,000	0/+?	< 100	1000s
IL	B, M, w	14/1994-97	6/1997	≥ 754	-	100s (?)	1000s
IN	M, w	-	-	-	-	200-300	100s-1000s
IA	B	10/1989-98	5/1998 ³	844 ³	+	-	
ID	B, m, w	11/1993	11/1993	1175-1401	+	50-150	NA
KS	B, M, w	6/1951-2000	3/1992-97	NA	NA	100s	10,000- 100,000
MB	B, M (?)	156/1979-91 (Manitoba)	33/1999 ⁴ (L. Win. only)	36,180 ⁴	NA	-	-
MI	B, M, w	48/1998	45/1997	31,079	0?	200	50,000
MN	B, M	75/1981-97	NA	NA ⁵	- /?	-	100s-1000s
MO	M, w	-	-	-	-	100s	1000s
MT	B, m	50-100/?	NA	NA	NA	-	NA
NE	B, M	14/1970-2000	14/1970-2000	NA	+/?	-	1000s
NM ⁶	B, M(?), W	3/1996	3/1996	130/1996	-/?	NA	NA
NY(inland)	B, M	12/1980s-97	12/1997	9,072 ⁷	+	-	1000s
ND	B, M	NA	NA	NA	+/?	-	NA
OH	B, M, w	2/1992-98	2/1998	1510	+	50 birds	25,000
ON	B	116/1980-97	74/1997	35,159	+	-	-
QC (sw)	B	5/1990-97	5/1990-97	212	+	-	-
SK	B	14/1980-91	10/1991	19,547	+	-	-
SD	B, M	36/1988-92	36/1988-92	> 2,962	-/?	-	NA
UT	B, m, w	8/1971-1999	8/1971-99	NA	NA	< 100	100s
VT	B	5/1982-99	2/1999	2886	+	-	-
WI	B	45/1980-97	23/1997	10,546	+	-	1000s
WY	B, M	25/1981-99	≤ 17/1994	NA	-/?	-	NA
Totals		490	217	≥ 153,056		1,000s	≥ 81,000

1 = B, Breeding; M, Migrant; W, Wintering; w, small numbers wintering

2 = For Alberta, we did not receive a completed survey. Only partial data available at this time.

3 = Number may be an underestimate, as not all known colonies were counted

4 = Number is for Lake Winnipegosis only; last province-wide survey was done in 1979, at which time 60 colonies were identified with 22,642 nests.

5 = No complete statewide census has ever been completed for Minnesota.

6 = Upper Rio Grande and Middle Pecos River Drainage; excludes colonies in Zone 2 (Table 1).

7 = Estimate for inland colonies only; est. pairs for entire state in 1997 = 12,675 (Miller 1997).

Figure 25. Distribution of Interior U.S. and Canadian DCCO breeding population (Zone 3), 1979 – 2000.

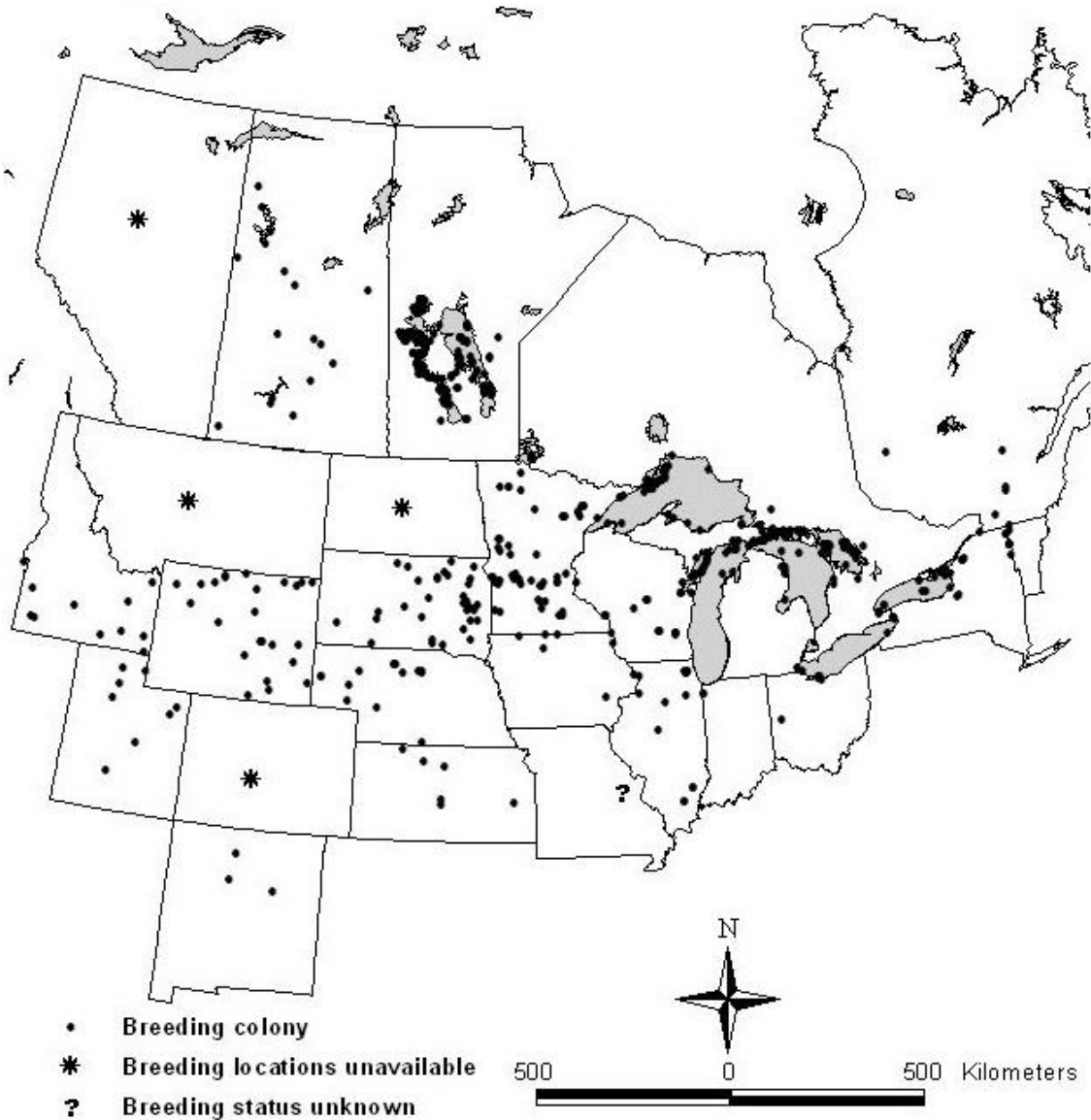


Figure 26. Distribution and size of active colonies in Saskatchewan during last complete survey (1991).

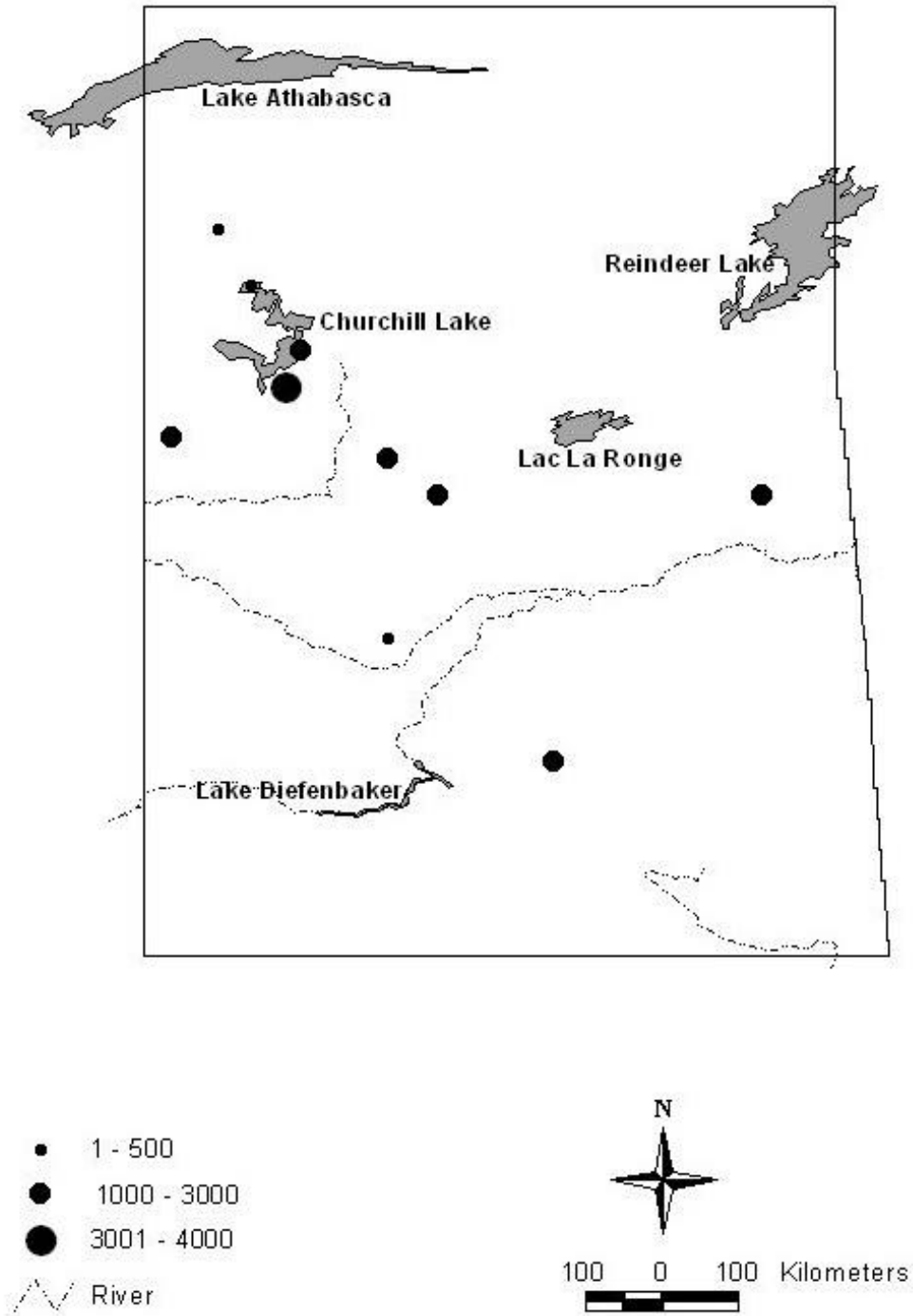


Figure 27. Distribution and size of active colonies on Lake Winnipegosis during last complete survey (1999).

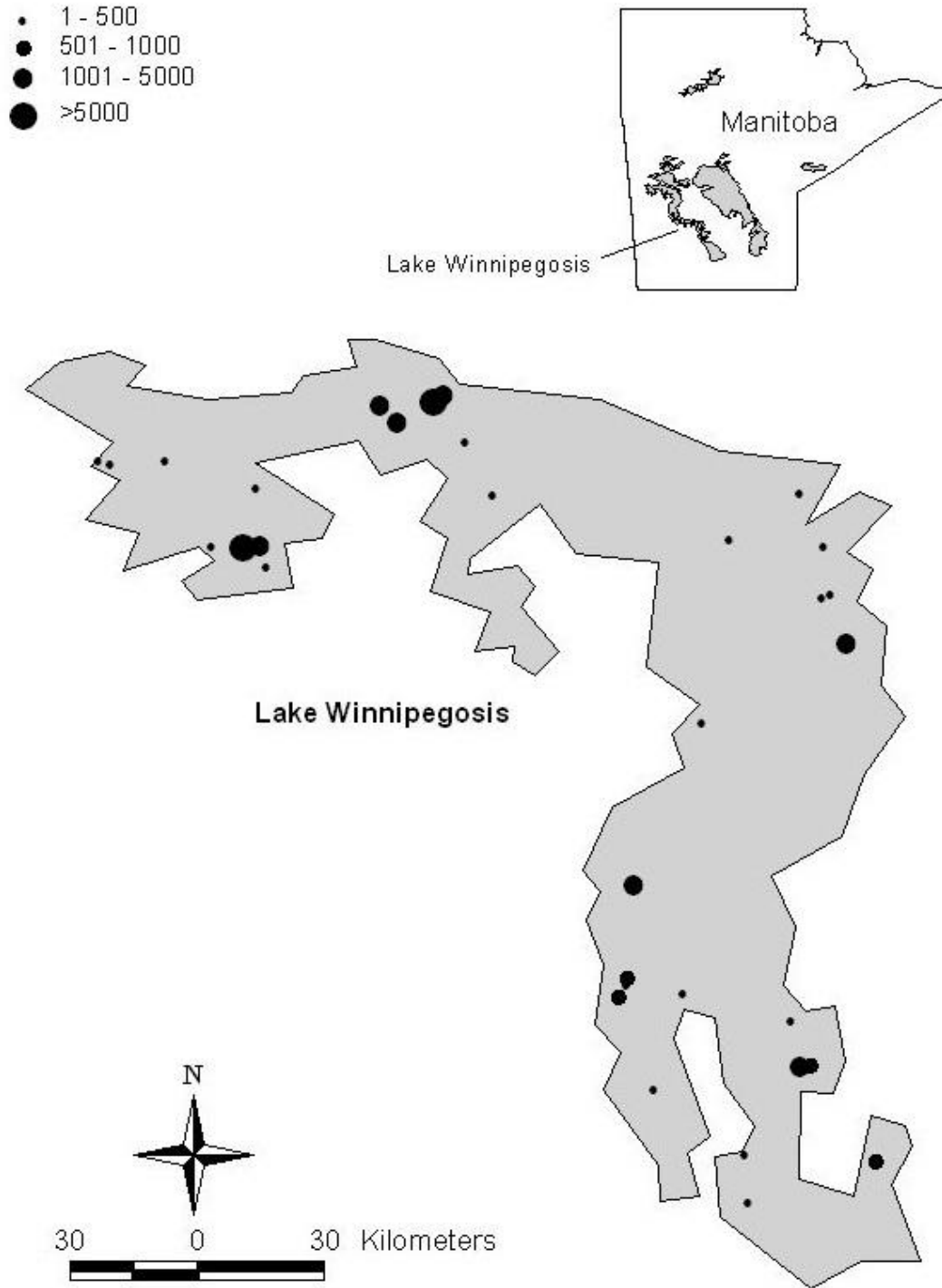
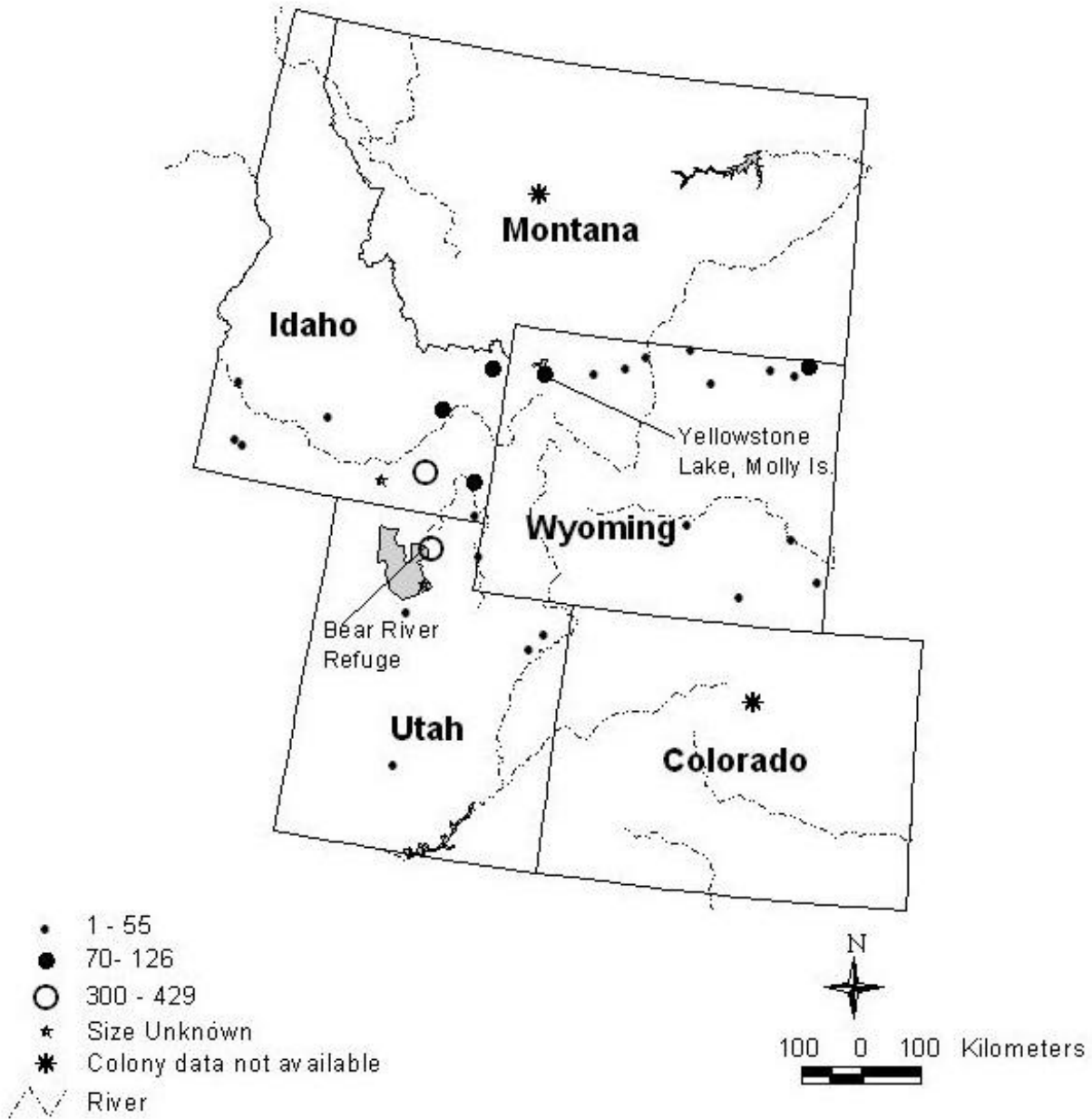


Figure 28. Distribution and size of active colonies in the Interior west-central states during last complete surveys (1986 – 1999)¹.



¹ Majority of colonies visited in 1990s; only 2 colonies with last visits in 1980s.

Figure 29. Distribution and sizes of active colonies in Interior mid-central states at time of last complete surveys.

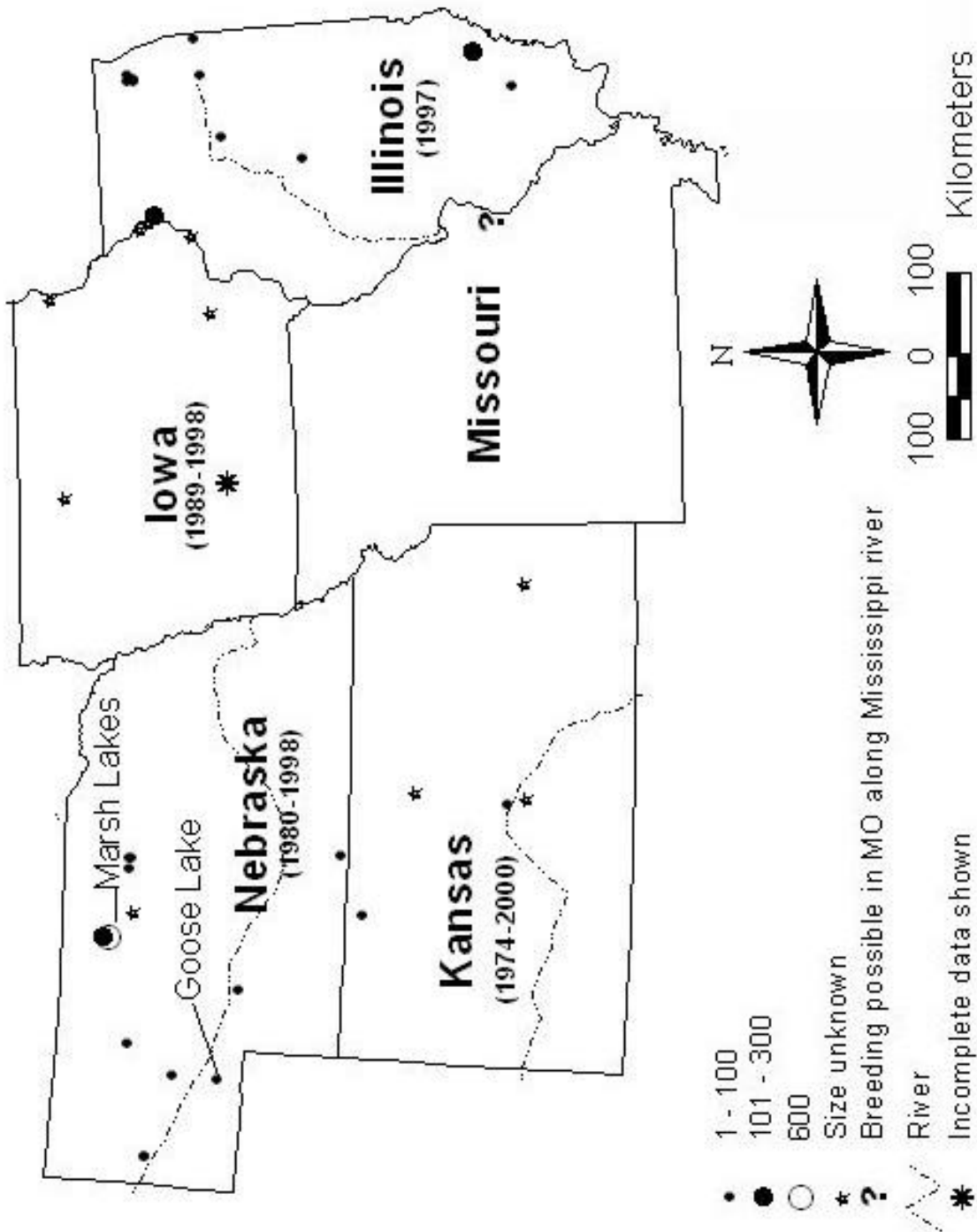


Figure 30. Distribution of active colonies (size data not available) in South Dakota and Minnesota (MN = 1981 – 1995; SD = 1988 – 1992).

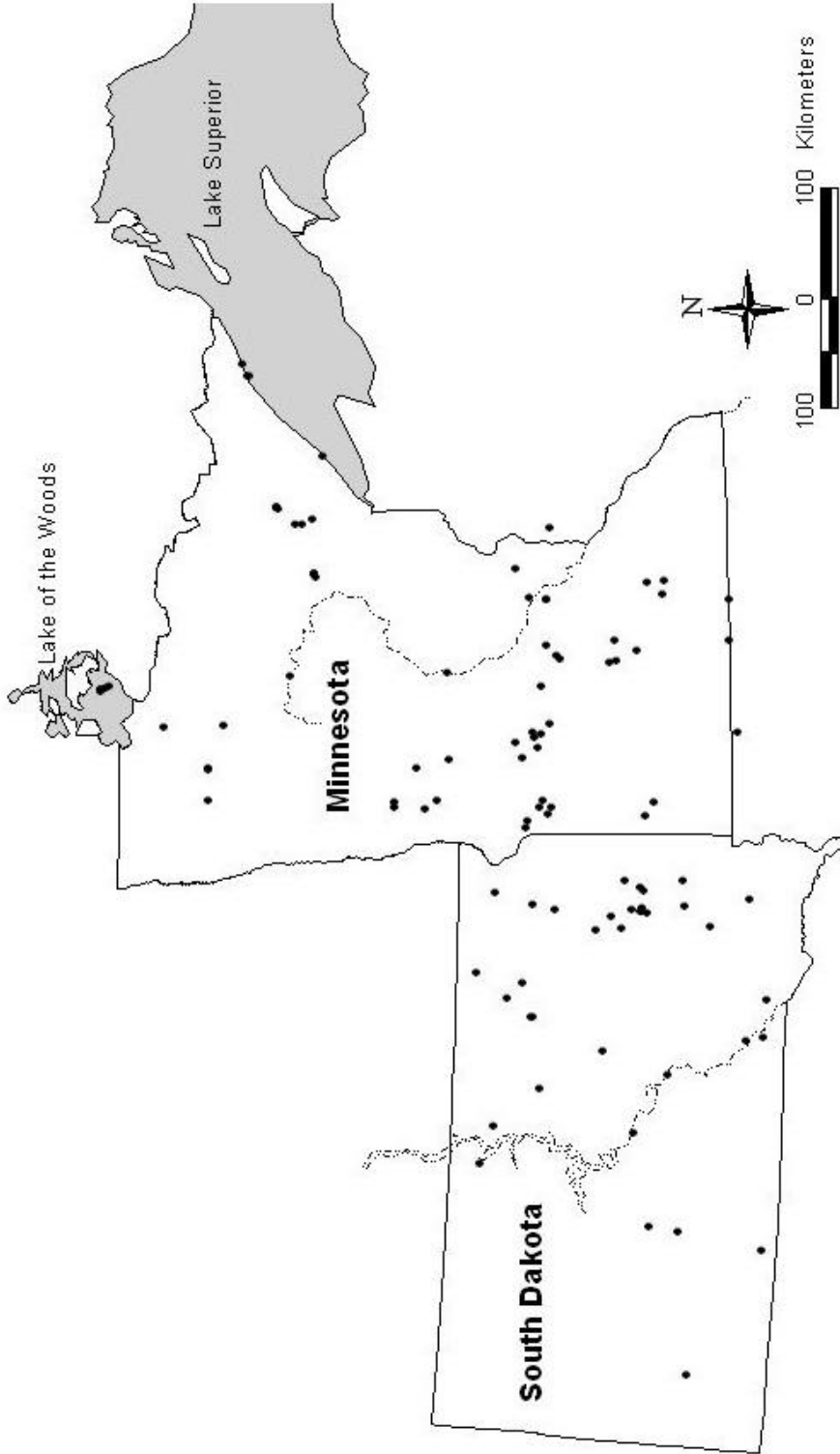


Figure 31. Distribution and size of active colonies in Wisconsin during last complete survey (1997).

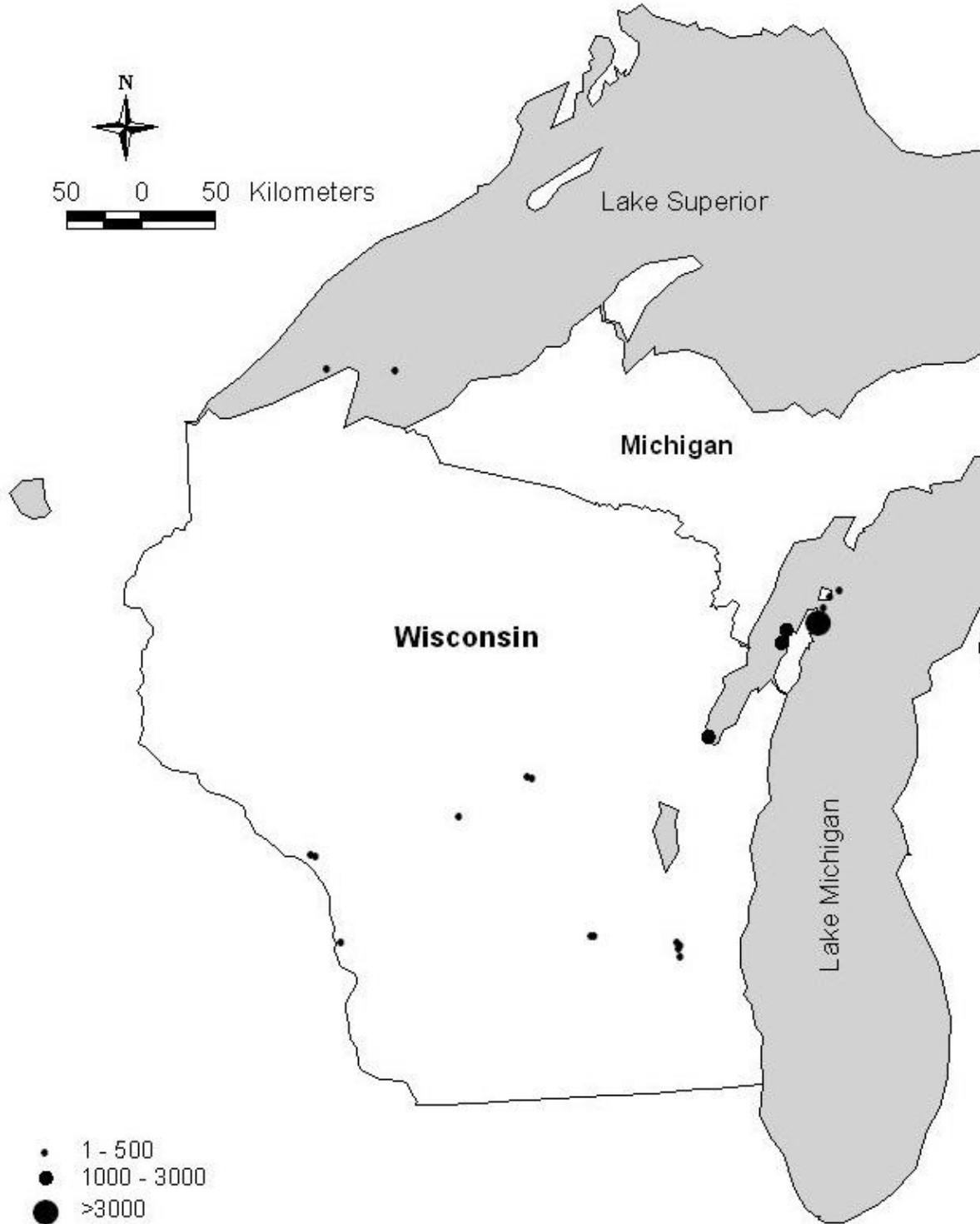


Figure 32. Distribution and size of active colonies in Michigan during last complete survey (1997).

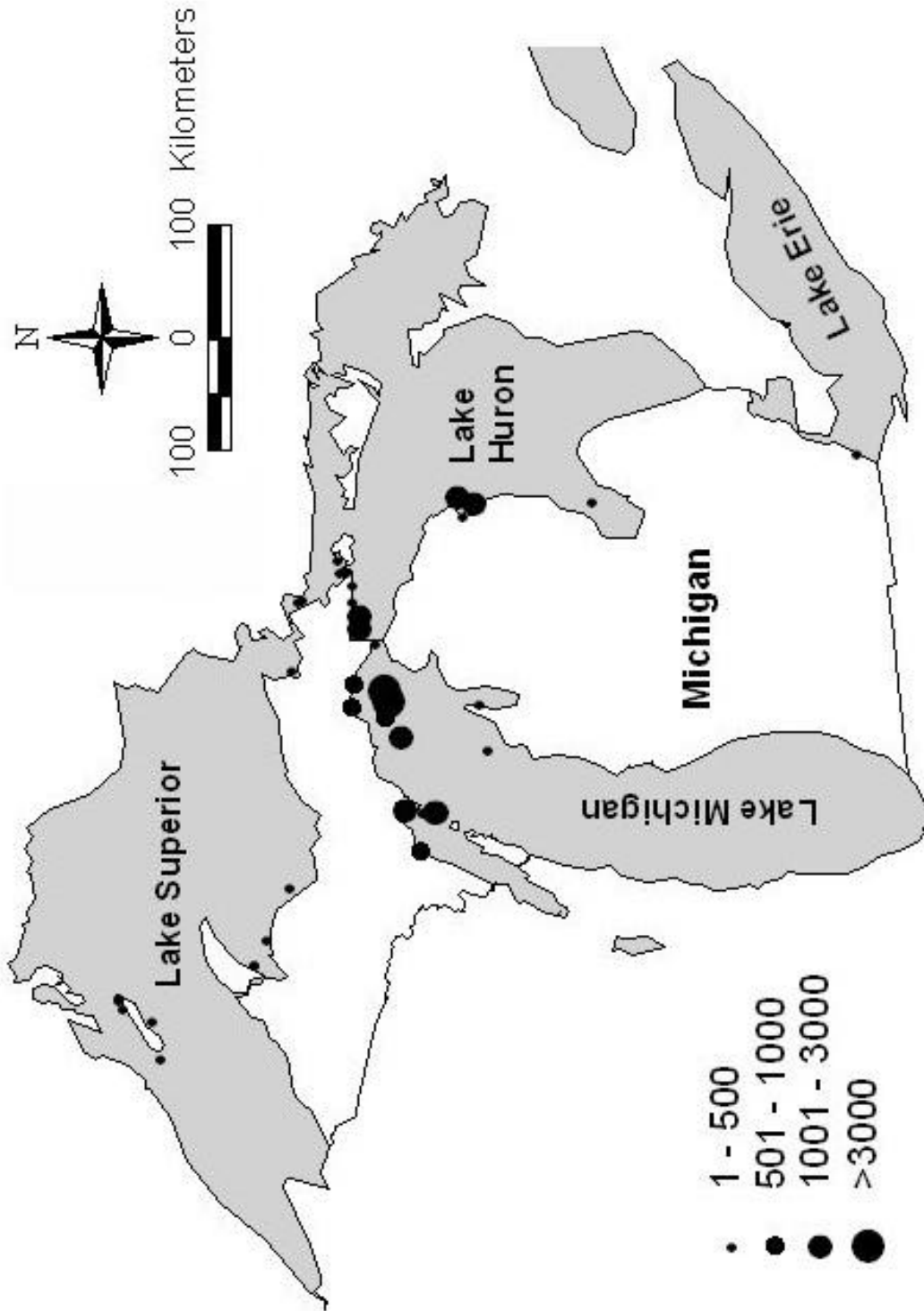


Figure 33. Distribution and size of active colonies in Ontario during last complete survey (1997).

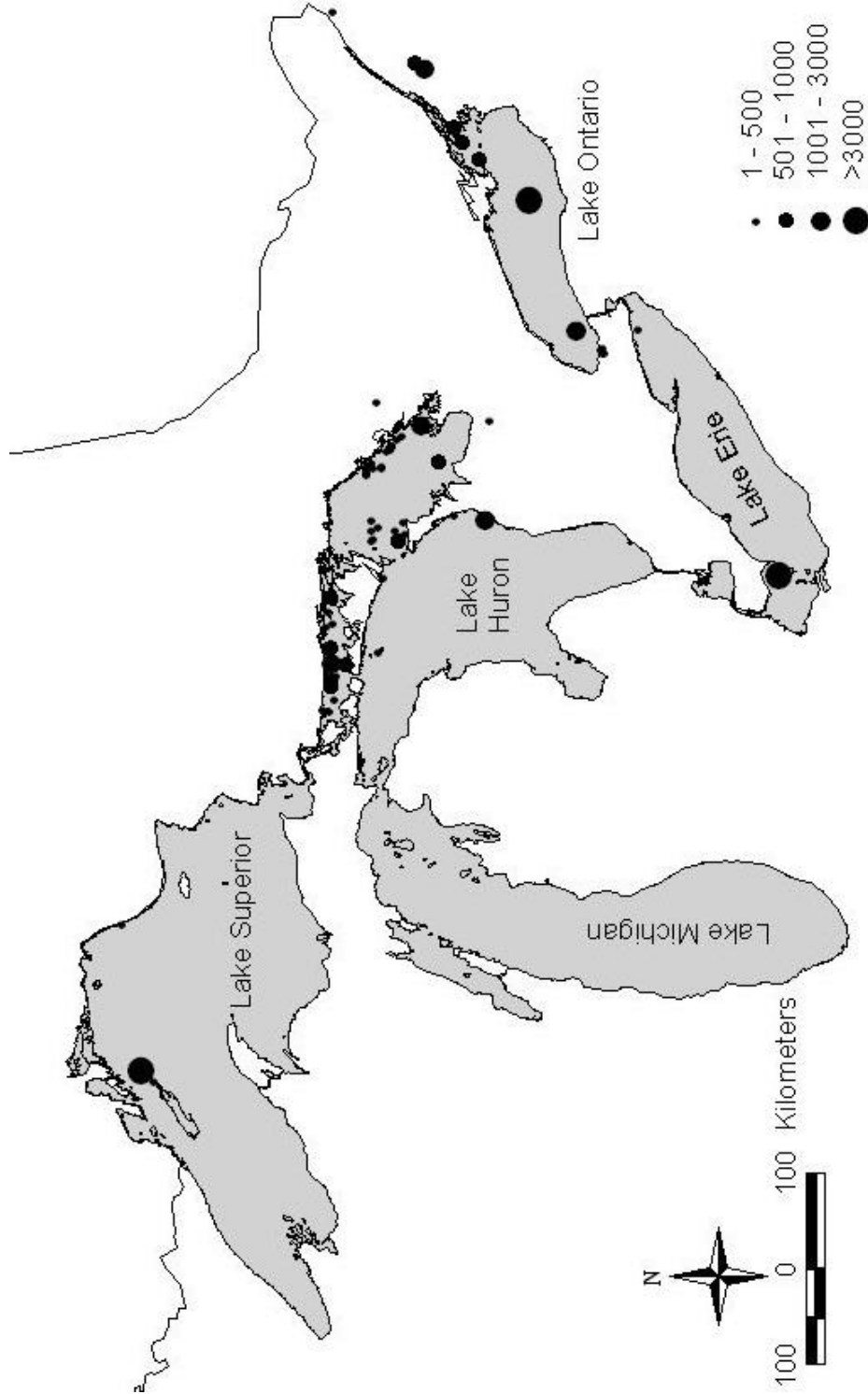


Figure 34. Detail of Lake Huron showing size and distribution of active colonies (1997).

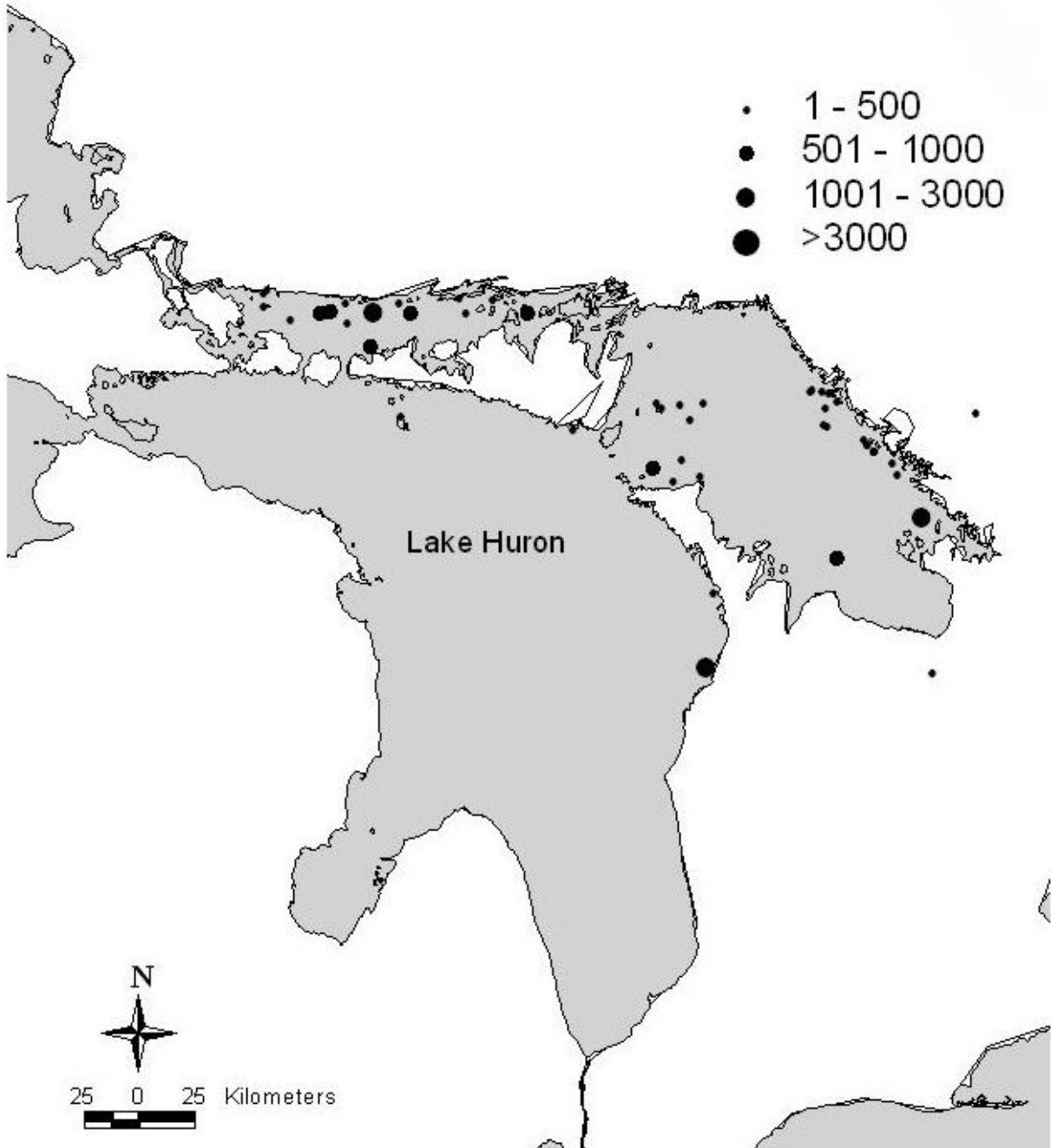
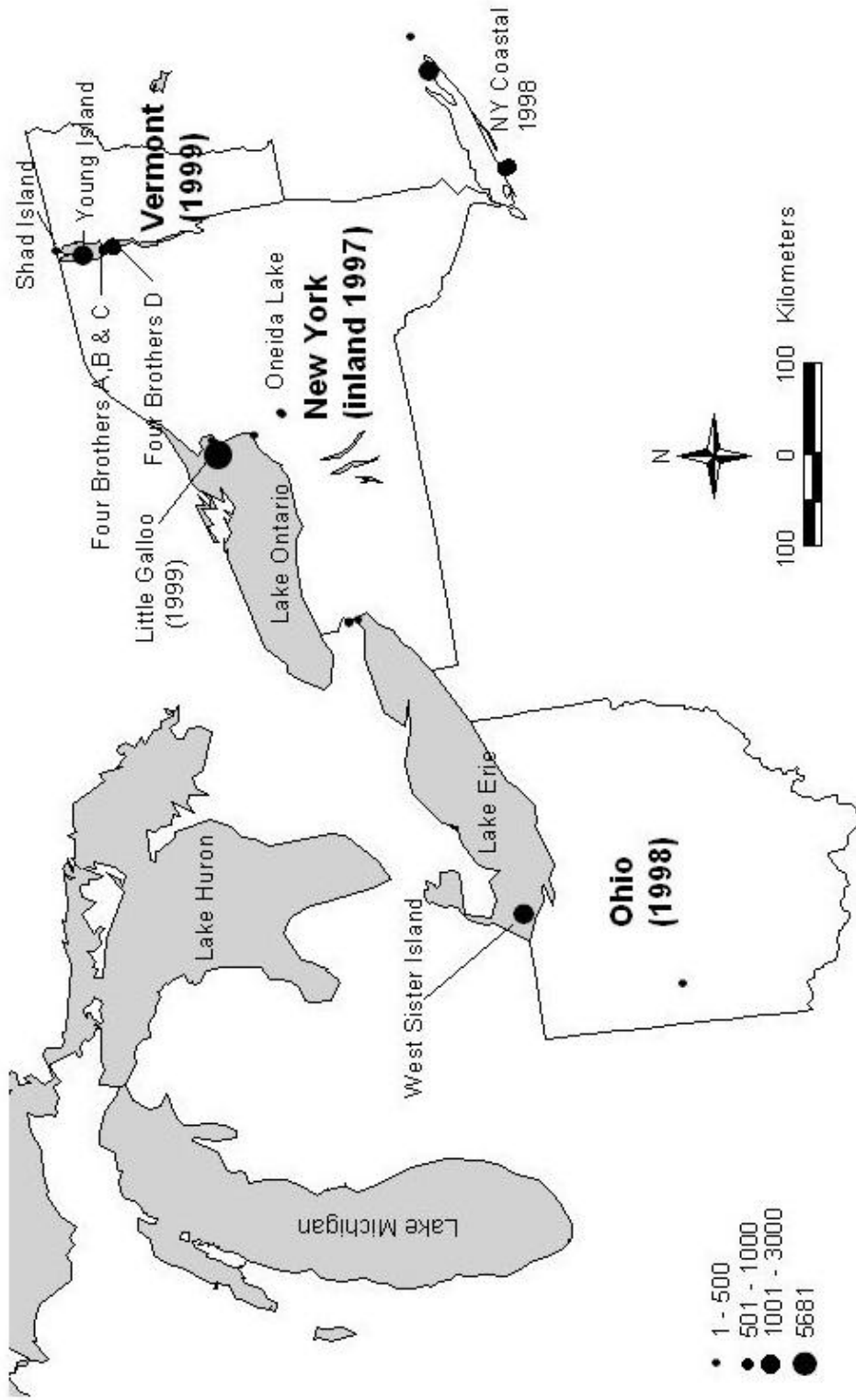


Figure 35. Distribution and size of active colonies in Ohio, New York, and Vermont.



Alberta

Summary of Population Data and Trends

(We did not receive a completed survey for Alberta.)

Korfanty et al. (1997) did not report breeding numbers for this province, but did state that the Double-crested Cormorant was added to the non-game list and removed from the endangered list in 1987, because numbers were increasing due to habitat protection.

Historical Information

In 1977, the species was designated as an endangered animal in Alberta due to population declines and habitat risks at nesting colonies.

No other information available.

Land Ownership

No information available.

Productivity

No estimates available.



Colorado

Summary of Population Data and Trends

(Surveys completed by K. Giesen, Colorado Division of Wildlife, T. Leukering and R. Levad, Colorado Bird Observatory.)

Breeding.

The DCCO has been documented as a nesting species in Colorado since 1931 (Kingery 1998). Beginning in 1987 and continuing through 1995, the Colorado Breeding Bird Atlas survey was conducted. Since the Atlas survey's inception, nesting has been documented at a total of 21 sites (R. Levad, pers. comm.), with about 1000 pairs estimated in the big mountain parks, along the Arkansas and Platte rivers, and on the northeastern plains (Kingery 1998). In 2000, a total of 28 nesting sites was recorded in the database. Of these, 19 were visited and 11 were active, with a total of 710 nests, but this was not a complete count (R. Levad, pers. comm.). Presently, precise locations and information on trends are not available. However, numbers of DCCO colonies and breeding pairs are known to have increased between the late 1930s and 1980s (Kingery 1998). In the 1980s the state population was estimated at 1000 pairs (Ryder 1996), and in 1992, 11 nesting colonies were reported (Andrews and Righter 1992). While numbers of colonies appeared to increase during the Atlas survey, numbers of breeding birds remained similar to those estimated in the 1980s (Kingery 1998). Construction of reservoirs is

presumed to be the main reason for increases and expansion (Andrews and Righter 1992). Location information may become available in 2001. Visits to all known colonies are planned for 2001, and annual counts will likely get underway (R. Levad, pers. comm.).

Winter.

In winter the DCCO is a very rare resident on eastern plains near foothills, and has also been recorded in western valleys in Mesa County (Andrews and Righter 1992). T. Leukering (pers. comm.) reported a few winter birds (< 10) at reservoirs in Boulder and Pueblo West, and K. Giesen (pers. comm.) reported < 20 birds overwinter in the state. Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 individuals were estimated during any count, but small numbers were observed at several sites.

Migration.

The species is reported as a common to abundant spring and fall migrant on eastern plains, with a high count recorded on Oct. 17, 1989, in Otero County (Andrews and Righter 1992). Peak numbers are observed in mid-October.

Historical Information

The earliest records we found of DCCOs in Colorado were those by Sclater (1912), who reported that only four records were verified up to 1912. The first breeding record was not obtained until 1931, when eight nests were found at Barr Lake.

By 1939, this colony had increased to 30 pairs. In 1995, 248 nests were reported at this site (Kingery 1998).

Land Ownership

No information provided.

Productivity

No information provided.



Idaho

Summary of Population Data and Trends

(Survey completed by Chuck Trost, Idaho State University.)

Breeding.

The DCCO has been documented as a nesting species in Idaho since at least 1941 (see Historical Information, below). Burleigh (1972) reported that 11 specimens collected from different locations in Idaho were examined to determine subspecies, and were found to be intermediate in their characters between *P. a. auritus* and *P. a. albociliatus*, but closer to *auritus*

(Burleigh 1972). The most recent survey for colonially nesting waterbirds in the state, conducted in 1993, documented 11 active colonies and estimated 1,175 – 1,401 pairs (Trost 1994). Prior to this, a survey for colonially nesting waterbirds conducted in 1984 found five colonies and estimated 805 – 880 pairs (Trost 1985). Peterson (1977) reviewed the distribution of colonial waterbirds in Idaho and reported six known DCCO colonies in the state, but did not report number of pairs. While numbers of breeding pairs and colonies appear to have increased since 1984, increases may be due, in part, to a better response to the questionnaire developed for the 1993 survey (Trost 1994).

Winter.

In winter, small numbers (40 – 50 birds) can be found along the lower Snake River (C. Trost, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified one site where ≥ 100 individuals were estimated: Hagerman Valley (129 in 1995). Data collected at this site in the 1990s indicate that numbers are slowly increasing, but analysis of recent data has not yet been conducted. Multiple sites with smaller numbers were also identified.

Migration.

No important stopover sites or significant concentrations of migrants were reported, though it does occur as a spring and fall migrant (C. Trost, pers. comm.).

Historical Information

The earliest documented nesting record appears to be from Bear Lake in southeastern Idaho, and was obtained in 1941 (Behle 1941). However, the bird was documented in the state much earlier than this. It was included in the first list of birds for the state. Lewis, in his account of the Lewis and Clarke expedition, recorded the species on the Clearwater River, October 8, 1805. Other early records of birds apparently in migration were obtained in the late 1800s in both the northern and southern portions of the state (Burleigh 1972).

Land Ownership

The following land ownership information was provided for breeding colony sites: Blackfoot Reservoir is probably owned by the Bureau of Reclamation (BREC), but the water belongs to the Bureau of Indian Affairs (BIA); Mud Lake WMA is state land managed by Idaho Fish and Game; Bear Lake, Minidoka and Deer Flats NWRs are federal land, managed by USFWS; the Mormorn Reservoir may be owned by BREC; Island Park Reservoir maybe owned by Henry's Fork Canal Co.; and American Falls Reservoir is managed by the BREC but the land is on BIA.

Productivity

No information provided.



Illinois

Summary of Population Data and Trends

(Survey completed by J. Herkert, Illinois Endangered Species Protection Board, Springfield, IL.)

Breeding.

The Double-crested Cormorant was first documented as a breeding species in Illinois by Ridgway (1874). Currently, the state has a relatively small number of nesting birds. In the 1990s, there was a total of 14 known colony sites, all at inland lakes, rivers and marshes. Eleven complete censuses of breeding colonies have been conducted between 1979 and 1997. Since 1979, significant increases have been reported, and in 1995, numbers peaked at 1077 pairs. Thereafter, numbers declined; in 1997, the last year for which survey data were available, ≥ 754 pairs were estimated at 6 colonies. Complete survey data were only available for 7 years between 1986 – 1997, when numbers increased at an average annual rate of 11.6 % (Figure 36).

Winter.

A small number of birds were reported to winter in the state (J. Herkert, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified three sites where ≥ 100 individuals were estimated: Horseshoe Lake (100 in 1993); Rend Lake (> 100 have been recorded on multiple counts in the 1990s, with a high of 1,597 individuals in 1998); and Pere Marquette Park (100 in 1998). Data were not consistent enough at the three sites combined to create a figure showing any potential trends. At the Rend Lake site large increases occurred in 1990, 1991, 1994 and 1998, and may indicate that this site is becoming a relatively significant wintering area for this region.

Migration.

Thousands are reported during the peak of spring migration, which occurs during March – April (J. Herkert, pers. comm.).

Historical Information

In the late 1800s, the Double-crested Cormorant nested near Mt. Carmel (Ridgway 1874), Lacon (Barnes 1890) and Peoria (Loucks 1893). Nesting was also recorded near Havana and Clear Lake in 1910, but these birds were severely persecuted (Lewis 1929). At this point, nesting appears to have ceased in the state or to have become very sporadic; Lewis (1929) was unaware of any nesting after this time. Bohlen and Zimmerman (1989) reported that the Havana colony, which had “several nests”, was shot out by fishermen in 1910, but that there were probably colonies elsewhere in the state. In 1960, the species was added to the Illinois endangered species list; at this time there was only one known nest site at Thomson, IL, on the Mississippi River with just seven nests. Artificial nesting platforms were later installed, and the colony at Thomson began to recover; 16 nests were recorded in 1978, and >110 were recorded in 1986. Bohlen and Zimmerman (1989) reported that very small colonies had also “recently” been found at Rend Lake, Lake Renwick, Putnam Co., and Bartonville; we assume these colonies were discovered in the late 1970s and early 1980s, because the IL DNR reported only 25 nests for 1979, the earliest year for which the DNR had state totals (though number excludes Thomson colony; J. Herkert,

pers. comm.). More small colonies continued to be discovered, and in 1997 there was a total of 14 known nesting sites.

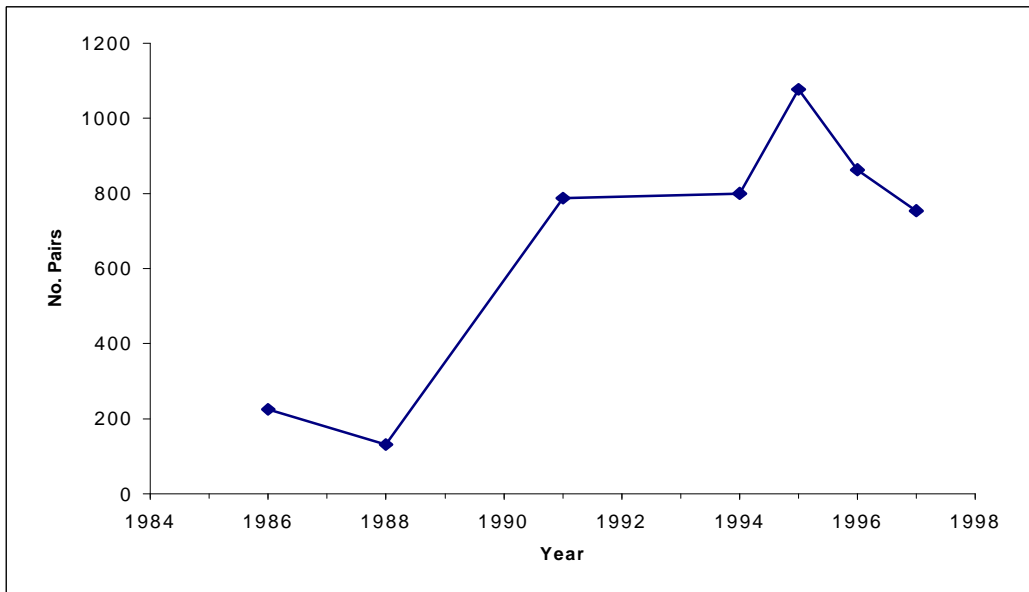
Land Ownership

Land ownership was provided for two colony and migration stopover sites, Carlyle Lake and Rend Lake. Both of these were reported as public lands. Carlyle Lake is federal land, owned by the Army Corps of Engineers; no additional information was provided for Rend Lake.

Productivity

No estimates available.

Figure 36. Changes in numbers of breeding pairs in Illinois, 1986 – 1997.



Indiana

Summary of Population Data and Trends

(Survey completed by J.S. Castrale, Indiana Dept Natural Resources, Mitchell, IN.)

Breeding.

Does not breed in the state. Although not reported by atlas workers or Breeding Bird Survey participants in Indiana, this species was as frequent as Common Loons on Summer Bird Counts, and recorded most commonly in northwestern Indiana (Castrale et al. 1998). Some recent observations of birds carrying nesting materials have been reported, but nesting has not been documented (J. Castrale, pers. comm.).

Winter.

In most years a small number of birds, 200 – 300, linger on into winter months (J. Castrale, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 individuals were estimated during any count, but small numbers were observed at some sites.

Migration.

During the peak of fall migration, September – October, up to 9000 birds have been reported. During the peak of spring migration, mid-April – early-May, smaller numbers ranging from 100 – 300 birds reported. Recently, records in all seasons have become more numerous and birds have been observed throughout the state (J. Castrale, pers. comm.).

Historical Information

The earliest record of the Double-crested Cormorant in Indiana dates from 1858 (West 1958), and Robert Ridgway thought the cormorant nested in Gibson and Knox counties prior to 1900 (Butler 1898). However, the first circumstantial evidence of nesting was not obtained until September 1929, when cormorants were present in “considerable numbers” at Hovey Lake and a nest was observed. Cormorants nested at this site until at least 1953 in small numbers (< 10 pairs). Nest building was also noted at Willow Slough Fish and Wildlife Area in 1953 (Mumford and Keller 1984). The last breeding record was obtained in the same year.

Land Ownership

Land ownership was provided for the 1 major stopover site reported, Hovey Lake FWA. This is state land, owned by the Indiana Division of Fish and Wildlife.

Productivity

Not applicable.

**Iowa*****Summary of Population Data and Trends***

(Survey completed by K. Bogenschutz, Iowa Dept Natural Resources, Boone, IA.)

Breeding.

The Double-crested Cormorant returned as a fairly regular breeder in Iowa in 1984 (see Historical Information, below, for breeding history), with nesting documented at Sabula in Jackson County, on the Mississippi River, and at the Coralville Reservoir in eastern-central Iowa. Over the last decade, nesting has been documented at a minimum of 10 sites. The Iowa DNR reported 8 colony sites on islands or lakes along the Mississippi River that were active between 1989 – 1998. Five of these colony sites were counted in 1998, and 844 pairs were

estimated. However, this may be an underestimate, as not all colony sites on the Mississippi were counted. Additionally, the Iowa DNR did not report the Coralville Reservoir site as one of the known breeding sites, though nesting was observed there in 1990. They also did not report nesting in north-central Iowa, which was documented in 1991 at the Union Slough National Wildlife Refuge in Kossuth County (Ehresman 1996). We were unable to obtain current information on the status of these colonies. Because Iowa has not conducted systematic statewide censuses of all known cormorant colonies, we estimated a population trend by examining changes in colonies that had been counted at least twice. Of the 10 colonies, five fit this requirement; three showed increases, while two showed decreases. Based on changes in these colonies, cormorants appear to be increasing in the state, which supports the opinion of the Iowa DNR (K. Bogenschutz, pers. comm.).

Winter.

Cormorants are not known to winter in the state (K. Bogenschutz, pers. comm.).

Migration.

No significant numbers of migrants were reported (K. Bogenschutz, pers. comm.).

Historical Information

The Double-crested Cormorant once bred abundantly in northern Iowa, but by the turn of the century, it occurred mainly as a migrant throughout the state (Cooke 1888; Anderson 1907; Lewis 1929). It was not recorded as a breeding species again until 1934 and 1936, when up to 12 nests were documented along the Mississippi River in Lee County. Between this time and 1984, scattered breeding records were reported along the Mississippi and Missouri Rivers, and at the Coralville Reservoir. The breeding colonies documented in 1984 were the first colonies reported since 1966 (Ehresman 1996).

Land Ownership

All colonies in Iowa are on federal lands (K. Bogenschutz, pers. comm.).

Productivity

At one of the colonies, Butler Lake, fledging rates (young / nest) were provided, and ranged from 1.25 the first year of breeding to 2.7 in 1994. In 1996, when the colony was estimated at 364 pairs, the fledging rate was 2.3.



Kansas***Summary of Population Data and Trends***

(Survey completed by W. Busby, Kansas Biological Survey.)

Breeding.

DCCOs have been documented as a nesting species in Kansas since at least 1951 (Thompson and Ely 1989). A total of six known colonies was reported between that time and 2000. The Kansas Breeding Bird Atlas, conducted 1992 – 1997, confirmed breeding at only three sites: Cheyenne Bottoms, Quivira National Wildlife Refuge and Wolf Creek Reservoir. Kansas does not conduct regular statewide surveys for DCCOs, and trend information is not available. However, breeding numbers are thought to be small (all colonies are always < 300 pairs) and use of individual breeding sites from year to year is erratic (W. Busby, pers. comm.). The largest colony reported was that at Glen Elder Reservoir in Mitchell County, which had as many as 220 pairs since 1974, but by 1989 there were many fewer (Thompson and Ely 1989).

Winter.

In winter DCCOs are not usually present, though a few sometimes remain until midwinter when there is no more open water (Thompson and Ely 1989). A few have also been observed in mild years. CBCs conducted 1959 – 1988 suggested an increase of 3.2 % per year (Sauer et al. 1996). Review of CBC data collected between 1984 – 1998 identified three sites where ≥ 100 DCCOs were estimated: Lawrence (150 in 1990, 216 in 1994); Waconda Lake (190 in 1998); and Webster Reservoir (110 in 1998). However, at the latter two sites CBCs were not initiated until 1992, and there were not enough consistent data for the three sites combined to create a figure showing any potential trends.

Migration.

Large numbers of migrants, 10,000 – 100,000, are estimated to occur, with most found at large Army Corps and Bureau of Reclamation reservoirs in the eastern 2/3 of the state, but this information has not been formally collected (W. Busby, pers. comm.). Migration peaks occur in mid-April and early October (Thompson and Ely 1989).

Historical Information

No information obtained.

Land Ownership

Land ownership information was provided for four breeding sites: the Kirwin and Glen Elder Reservoirs are both owned by the Army Corps of Engineers, the Cheyenne Bottoms colony is on state land, and the Norton Reservoir colony is on Bureau of Reclamation land. Kirwin, Glen Elder, and Norton are all federal reservoirs owned and operated by the Bureau of Reclamation.

Productivity

No information provided.



Manitoba

Summary of Population Data and Trends

(Survey completed by W. Koonz, Manitoba Dept. of Nat Res, Winnipeg, Manitoba.)

Breeding.

Manitoba has the largest number of breeding Double-crested Cormorants in North America. Birds nest on shallow lakes throughout the southern half of the province and on lakes Manitoba, Winnipeg, Winnipegosis and in the central portion of the province. Because Manitoba is so large and has so many colonial waterbird breeding sites, province-wide surveys are very difficult to accomplish in one season. The last province-wide survey was conducted in 1979 over a four day period with a fixed wing aircraft; numbers counted in this survey probably underestimated the breeding population (B. Koonz, pers. comm.). Because complete province-wide surveys have not been regularly conducted, province-wide population estimates and trends are not available. However, data collected from colonies over the last 20 years provide some information on population changes. Between 1979 – 1991, breeding was documented at approximately 156 sites; of these, 26 sites were counted at least twice, and 19 showed increases. The province-wide survey conducted in 1979 counted 22,642 active nests at 60 colonies; 9053 nests were at 17 colonies on Lake Winnipegosis (Koonz and Rakowski 1985). In 1987, 35,181 nests were reported at 37 colonies on Lake Winnipegosis alone, suggesting a dramatic increase over previous Lake Winnipegosis and province-wide numbers (Hobson et al. 1987). The population at Lake Winnipegosis continued to increase until at least 1989, when a high of 51,788 pairs on 40 islands was estimated (B. Koonz, pers. comm). Lake Winnipegosis was censused again in 1999, at which time 36,180 nests were counted on 33 islands. (Figure 37).

Most of the growth on Lake Winnipegosis appears to have occurred during the 1980s, when the average annual growth rate was 19 %. We did not calculate an average annual increase rate based on the 1999 survey data for the following reasons. While breeding numbers appear to have declined since the late 1980s, this decline is probably not significant. During 1999 surveys, at least 1000 dead birds that had been shot were observed at several colonies, and many non-breeding birds were also observed. Active shooting and harassment of birds occurred at one colony even while surveyors were trying to obtain nest counts. Because human disturbance was so prevalent, many birds were probably not breeding, possibly explaining why fewer nests were counted. The number of birds breeding on Lake Winnipegosis, overall, has probably remained fairly stable over the last decade (B. Koonz, pers. comm.).

One important observation from surveys done in the late 1980s and in 1999 at Lake Winnipegosis was that a large proportion of cormorant nests was found in trees on islands not traditionally used for nesting by cormorants. Historic records from Bent (1922) and McLeod (1943) consider cormorants to nest exclusively on the ground. Vermeer (1973) also noted that cormorants nested in trees in the prairie provinces only when there were no treeless islands in the vicinity. Establishment of new colonies on forested islands may be due to increases in cormorant numbers and / or disturbance of traditional cormorant nesting islands (Lewis 1929; Vermeer

1969, 1973; B. Koonz, pers. comm.); the forested islands colonized are usually distant from major travel routes where unofficial cormorant control frequently occurs (B. Koonz, pers. comm.).

Winter.

No birds are known to winter in the province (B. Koonz, pers. comm.).

Migration.

The peak period of migration occurs in late August – early September, when birds leave for southern wintering grounds (B. Koonz, pers. comm.).

Historical Information

The Double-crested Cormorant has been an established breeder in Manitoba for well over a century. Lake Winnipegosis was recognized for its cormorant colonies just prior to 1900 (Koonz and Rakowski 1985), and Bent (1922) remarked that he had not seen greater densities of Double-crested Cormorants than on this lake. Lewis (1929) noted early records (1891 – 1909) of cormorants nesting at Shoal Lake, lakes north of Touchwood Hills, Ossowa, and Selkirk Settlement, but cormorants appeared to have ceased nesting at these places by the time Lewis was writing. Mendall (1936) recorded a total of 9,320 nests from at least 27 islands on seven Manitoba lakes.

Koonz and Rakowski (1985) and Hobson et al. (1987) described the history of the Double-crested Cormorant in Manitoba during the twentieth century, and that information is summarized here, unless otherwise noted. As in many areas of the continent, commercial fishermen blamed cormorants for losses in Manitoba's commercial fishery, and from 1943 – 1951, the Manitoba government conducted a cormorant control program on Lake Winnipegosis. Human disturbance reduced numbers on this lake from 9,862 nests in 1945 to 4,656 nests in 1951. However, even after government control stopped, declines continued on Lake Winnipegosis through the 1960s, and by 1969, there was only a total of 1,403 nests counted. These declines were attributed to large scale destruction of eggs and young birds by people (Vermeer 1973).

In 1963, the species became officially protected under the provincial Wildlife Act. In 1969 Vermeer (1969) recorded 4772 nests counted from 37 islands on 13 of Manitoba's lakes. Unofficial destruction of eggs, nestlings and adults, combined with the effects of DDT on nest success, probably contributed to the post-government control decline. During the 1970s and early 1980s, numbers began increasing again, likely in response to the ban on DDT. Persecution by humans continues, apparently with little substantial impact to overall numbers, but substantial changes in Manitoba's population distribution may result (B. Koonz, pers. comm.; see Impacts to Avian Species section).

Land Ownership

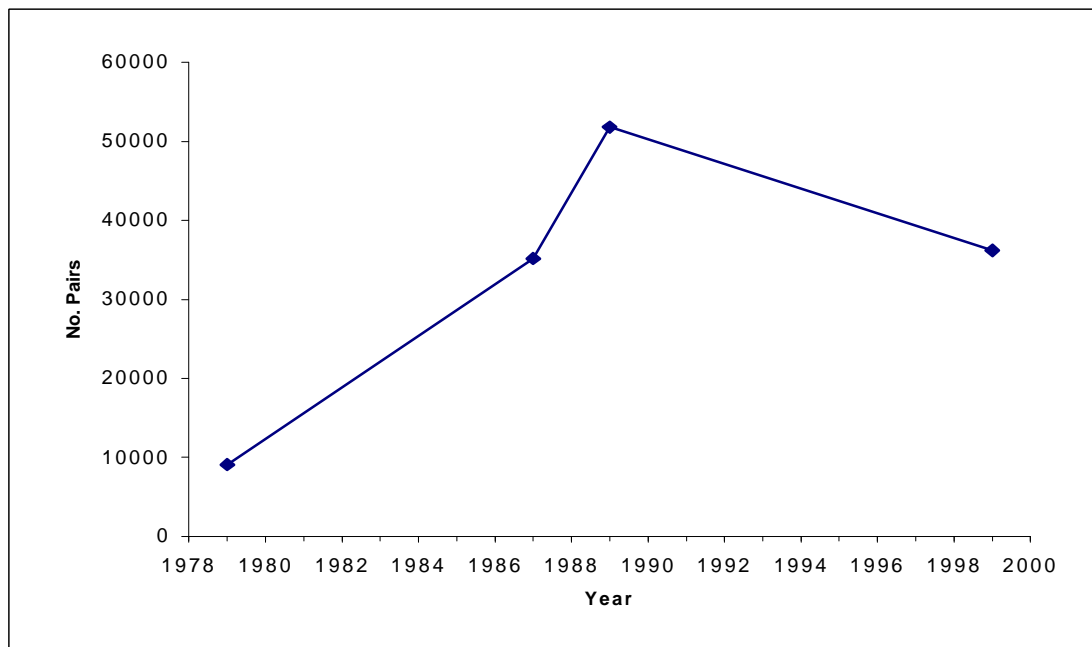
Most if not all Manitoba islands used by cormorants for breeding are Crown owned (B. Koonz, pers. comm.).

Productivity

Hobson et al. (1987) provided productivity estimates for 32 colonies on Lake Winnipegosis in 1987. Productivity was defined as number of mobile chicks per initiated nest in July and August. Thirteen colonies showed evidence of human disturbance; the mean

productivity at these colonies was approximately 0.4 chicks per pair (range 0 – 1.2). Nineteen colonies were undisturbed, and mean productivity at these colonies was 1.8 chicks per pair (range was 0 – 4.0; the 0 was at a colony that may have been disturbed by humans or a storm).

Figure 37. Changes in breeding numbers on Lake Winnipegosis, 1979 – 1999.



Michigan

Summary of Population Data and Trends

(Survey completed by J.P. Ludwig, The Sere Group, Kingsville, Ontario; additional information provided by Glenn Belyea, Michigan Dept Nat. Resources, E. Lansing, MI.)

Breeding.

Cormorants disappeared from Michigan as a breeding species in the 1960s; at the end of the 20th century, however, Michigan has by far the largest number of breeding cormorants in the U.S. Great Lakes, and constitutes one of the major breeding sites in North America. The species resumed nesting in the state in 1977, when nine nests were counted at Gravelly Island. Since that time, two more complete censuses of the Great Lakes have been conducted (1989, Scharf and Shugart 1998; 1997, Cuthbert, unpublished data), and 45 breeding sites were documented in 1997. In 1998, three more breeding sites were discovered, bringing the total number of known breeding sites in Michigan to 48. Ludwig (1991) reported 1 inland colony site, but status of this site was not determined for this assessment. Between 1977 and 1997 numbers increased at an

average annual rate of 50 % (Figure 38). However, the rate of increase has slowed; between 1989 – 1997, the average annual rate of increase was 27.8 %, compared to 67.4 % during 1977 – 1989. The current rate of increase is not known, but in 2000 D. Trexel censused Michigan colonies (with the exception of three sites) and found that overall growth rate for these colonies, while still positive, had declined significantly. In addition, only one new colony site was found. A few concurrent years of data would help determine by how much growth has slowed and whether cormorants are approaching carrying capacity in this state.

Winter.

A small number of birds winter in the state (J. Ludwig, pers. comm.).

Migration.

During peak migration periods, April and August, estimates of 50,000 birds are reported (J. Ludwig, pers. comm.).

Historical Information

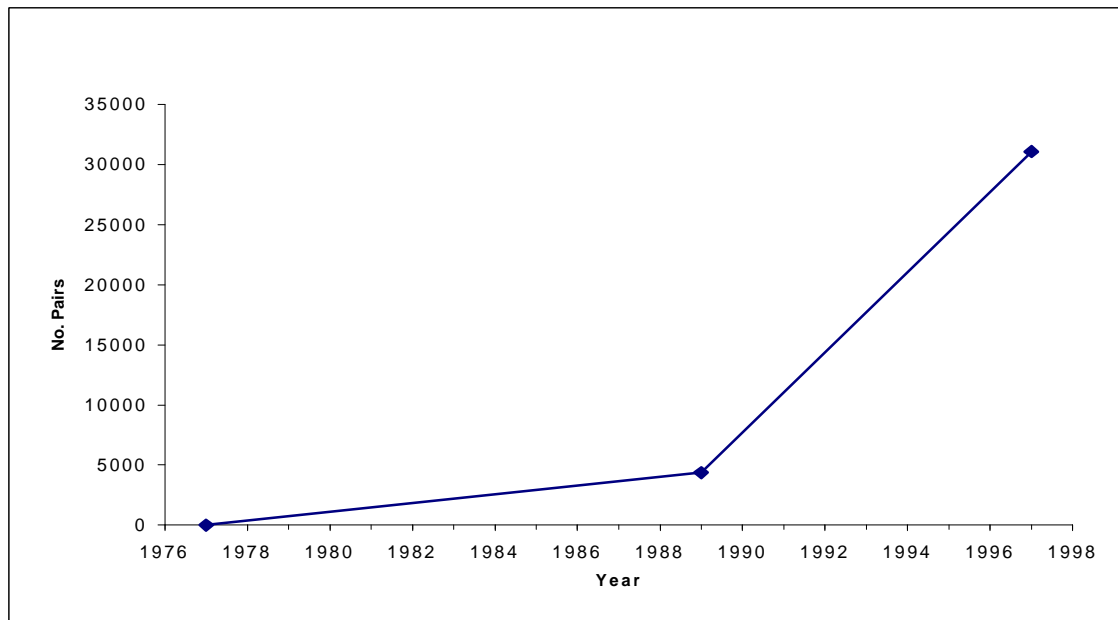
Changes in cormorant historical distribution and numbers within Michigan have been described by Ludwig and Summer (1995) (unless otherwise noted), and are summarized here. Small numbers of migrants (<20) were reported in the late 1800s during spring and fall migrations (Barrows 1912; Wood 1950). Given distribution known in the state in the late 1800s and early 1900s, Barrows (1912) speculated that cormorants might nest about the Great Lakes, but no records for Michigan were obtained until the 1930s. F.E. Ludwig (unpublished data) accumulated occasional breeding records at islands near Alpena (Lake Huron) between 1932 and 1941, and at Pismire, Hat and Bellows islands (Lake Michigan) in the 1930s; the population in the 1940s is estimated to have been between 200 – 500 pairs. Persecution by commercial fishermen and predation are thought to be responsible for regulating the population at these low levels. By the outbreak of WWII, creation of artificial habitat, such as flooded inland impoundments, provided breeding refugia for cormorants. However, DDT came into widespread use in the late 1940s, and by 1960, Great Lakes cormorant populations were in rapid decline due to DDE-mediated eggshell thinning. The species vanished from all Michigan Great Lakes sites by 1960, and from inland impoundments by 1964. Double-crested Cormorants did not breed in Michigan again until 1977. Since that time, the species has undergone a phenomenal increase in this state.

Land Ownership

Of the 51 known colonies, ownership was identified at only 14. Of these, three (about 21 %) were on private lands, and the rest were on federal lands. One of the colonies was on private land, a site soon to be under federal ownership.

Productivity

No estimates of productivity were available.

Figure 38. Changes in numbers of breeding pairs in Michigan, 1977 – 1997.

Minnesota

Summary of Population Data and Trends

(We did not receive a completed survey for Minnesota; information on breeding numbers and locations was provided by T. Roden, MN Dept Natural Resources, Natural Heritage Program, St. Paul, MN and by J. McKearnon, Univ of Minnesota, Dept. Fisheries and Wildlife, St. Paul, MN.)

Breeding.

The Double-crested Cormorant has been documented as a nesting species in Minnesota since 1875, when full-grown young were observed in treetops at Lake Minnetonka (Roberts 1932). A total of 75 nesting sites were recorded between 1981 – 1997. Only three of these are on the Great Lakes (Lake Superior); the rest are all on inland lakes and marshes. Between 1977 – 1997, the Minnesota Department of Natural Resources conducted annual counts of colonial waterbird colonies, but because it has never conducted complete statewide censuses, total numbers for the state during any one year are not available. However, efforts have been made to consistently count the large or major colonies. McKearnon (1997) estimated population trends of cormorants in Minnesota by using a Breeding Bird Survey analysis method, in which each colony was treated as a route. Based on this analysis, McKearnon reported that cormorants increased in the state from 1981 – 1990 at an estimated rate of 20 % per year, and then declined at a rate of – 8 % per year until 1995. A few more years of data will help determine whether

cormorants are past their peak and are declining or fluctuating around an asymptote. In 1997, 13 colonies were counted and there was a total of 3943 pairs, compared to a total of 5059 pairs counted at 22 colonies in 1995.

Because Minnesota is one of the states where the Depredation Order is in effect, a higher priority should be placed on monitoring numbers in this state. Complete state-wide surveys of breeding birds should be conducted so that numbers of colonies and breeding pairs can be accurately assessed. Without this type of information, impacts of and to the birds and effect of the depredation order in terms of buffering DCCO impacts cannot be determined.

Winter.

No birds are known to winter in the state.

Migration.

During migration, peak flock size is usually 200 – 500 birds, though 1000 were reported in Oct, 1983. Peak spring migration usually occurs in the second half of April; peak fall migration occurs during the third week of October (Janssen 1987).

Historical Information

Lewis (1929) notes that Minnesota once undoubtedly contained a large number of Double-crested Cormorant breeding colonies, with nesting documented at Lake Minnetonka, Kawishiwi Lake, Lake of the Woods, Elbow Lake, Lake Shetack, Dead Lake, Heron Lake, Loon Lake and Lake Andrew, near Lanesboro and Faribault, and probably in many other places (Lewis 1929). Many of these colonies were described as “large”; at Loon Lake, hundreds of birds nested on the ground and in trees, and when the island was visited in the late 1800s, it was “covered” with nests (Roberts 1932). Huge numbers of birds were also observed during the migration around the turn of the century; Sennett (1891, cited in Lewis 1929) reported a flock of cormorants passing through the state that was “4 miles long and 1½ miles wide”. However, congregations of large numbers in the spring provoked professional fisherman and anglers to shoot adult DCCOs and destroy nest sites. In April, 1918, the Game and Fish commissioner received numerous complaints that $\geq 10,000$ cormorants were in the bottom-lands of the Minnesota River and were going out daily to the neighboring lakes to feed upon the fish; requests were made to drive away or destroy the birds. Roberts (1932) noted that the number of birds that remained in the spring to nest was growing steadily less each year, chiefly because of human persecution. By the mid 1920s, cormorants were known to breed at only Lake Minnetonka, Kawishiwi Lake, and Lake of the Woods, and they were rapidly disappearing from these sites (Roberts 1932). Additionally, Lewis (1929) reported the huge flocks observed by Sennett (1891) during migration were no longer seen. However, large numbers were still at least occasionally observed in the region through the mid 1920s (see Wisconsin summary).

Janssen (1987) states that migrants were “abundant” in the state until the 1950s; however, he reports flocks of only 1,000 – 5,000 birds, nowhere near the historic highs reported around the early part of this century. Numbers of migrants declined between the 1950s – 1980s, and then began to increase again; however, peak flocks reported in the 1980s consisted of only 200 – 500 birds (Janssen 1987). While current numbers of breeding pairs in the state appear to be abundant, it is not possible to compare these numbers with earlier ones reported around the turn of the century, because we do not have nest counts for earlier colonies.

Land Ownership

Land ownership was available for only two Great Lakes sites. Both of these colonies were on federal lands owned by the BLM.

Productivity

No estimates available.

**Missouri*****Summary of Population Data and Trends***

(Survey completed by J.D. Wilson, Missouri Department of Conservation.)

Breeding.

The status of the Double-crested Cormorant in Missouri is not clear. J. Wilson (pers. comm.) reported that it does not breed in the state, but Robbins and Easterla (1992) reported that some breeding may occur along the Mississippi River.

Winter.

Generally, cormorants do not winter in Missouri, except for small numbers that may remain on the larger southernmost lakes in mild winters (J. Wilson, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified three sites where ≥ 100 individuals were estimated: Springfield (100 in 1991); Montrose Lake Wildlife Area (104 in 1997); and Horton-Four Rivers (988 in 1998). At the first two sites DCCOs have been observed during the 1984 – 1998 time frame fairly consistently. At the Horton-Four Rivers site, counts were not conducted until 1995; DCCOs were recorded in 1996, when one bird was observed. Data were not consistent enough at the three sites combined to create a figure showing any potential trends.

Migration.

Several thousand pass through during spring and fall migration; peak numbers occur in April and October (J. Wilson, pers. comm.). The species is more common throughout the state during the fall migration, and high counts of 2000 + birds have been recorded (Robbins and Easterla 1992).

Historical Information

Early in the century, Widmann (1907) reported that the Double-crested Cormorant bred in “considerable numbers” in the Mississippi Lowlands. However, the species apparently declined as a breeder, and Bennitt (1932) listed it as an “uncommon breeder”. Robbins and Easterla (1992) note that it is not clear if Bennitt’s statement was based on current information at

the time or whether it was a reiteration of Widmann's observations. The last documented nesting in the state occurred in 1956 (Robbins and Easterla 1992).

Land Ownership

Five major stopover sites (roosting and feeding) were reported, and land ownership was provided for all of them. Four of the sites were on federal land, owned by the Army Corps of Engineers and USFWS; the other site was on state land, owned by the Missouri Department of Conservation (J. Wilson, pers. comm.).

Productivity

No estimates of productivity were available.



Montana

Summary of Population Data and Trends

(Survey completed by D. Flath, Montana Fish, Wildlife and Parks.)

Breeding.

We were not able to determine when the DCCO was first documented as a breeding species in Montana. Currently 50 – 100 colonies are estimated to occur statewide, but colonies are not monitored. Colonies probably range in size from 15 – 120 pairs; average colony size may be around 40 pairs (D. Flath, pers. comm.). No information is available on population trends. However, based on aerial observations, the species appears to be increasing along the Yellowstone River where heron colonies occur (D.Flath, pers. comm.).

Winter.

In winter, the species rarely occurs; there are only two known winter records (D. Flath, pers. comm.).

Migration.

No significant concentrations of migrants or stopover sites were reported (D. Flath, pers. comm.).

Historical Information

Very little information obtained. The species was recorded as a rare migrant along the large rivers in the eastern part of the state by Saunders (1921). This observation was based on records of migrants collected in the late 1800s and early 1900s.

Land Ownership

No information provided.

Productivity

No information provided.

**Nebraska*****Summary of Population Data and Trends***

(Survey completed by J. Dinan, Nebraska Game and Parks Commission.)

Breeding.

The DCCO has been documented as a nesting species in Nebraska since 1930 (Farrar 1997), but likely occurred as a nester earlier than this (see Historical Information, below). Between 1970 and 2000, a total of 14 colonies were known to be active. Regular statewide surveys for DCCOs are not conducted, so information on population trends is not available. However, in recent years, the number of colonies in the state appears to have increased. Most of these are small (< 100 pairs); only a few have persisted for many years. The apparent increase in number of colonies is likely a re-colonization of old breeding grounds (Farrar 1997). The two largest and most persistent colonies in recent years have been on islands in the Crescent Lake and Valentine national wildlife refuges. During the 1950s and 1960s, Canada Goose nesting structures were placed at both these refuges in an effort to restore Canada Geese as breeding birds to the Sandhills region. DCCOs took advantage of these structures and fairly large colonies developed at Goose Lake in the Crescent Lake NWR and at Marsh Lakes in the Valentine NWR. In 1991 the island at Goose Lake was inundated, and the number of nests declined (Figure 39). At the Marsh Lakes colony the number of nests fluctuated from a low of 240 to a high of 922 with lake levels; in 2000 this colony was estimated at 840 pairs (Figure 40) (J. Dinan, pers. comm.; Farrar 1997).

Winter.

In winter, DCCOs are usually not present, though during very mild winters when some lakes remain open individuals can be observed occasionally (Sharpe et al. 2001).

Migration.

In spring and fall, the DCCO is an abundant and regular migrant statewide. Peak concentrations are observed in April and October. During migration DCCOs utilize medium to large sized lakes and reservoirs, and less frequently rivers such as the Platte and Missouri (Sharpe et al. 2001). High counts ranging from 350 – 1500 individuals were made between 1994 and 1998 at Medicine Creek and Harlan Reservoirs, the North Platte NWR, Branched Oak Lake, Nebraska City and Holmes Lake, Lincoln. In 1999, a high of 5000 + individuals was observed at Harlan County Reservoir.

Historical Information

The first nesting record was not obtained until 1930 (Thorp Lake), but breeding was suspected in the early 19th century. Ducey (1988, 2000) described the species as a likely historic breeder in western Nebraska, and noted that it was a potential breeder as early as 1820 at the Engineer Cantonment area, Washington County. At this time it was a regular migrant along the Missouri River, and Coues (1874) listed it as a breeder in this area.

Land Ownership

Land ownership was provided only for the two large colonies at the Crescent Lake and Valentine NWRs; both are public lands administered by the USFWS (J. Dinan, pers. comm.)

Productivity

Data were provided on the number of fledged chicks at Valentine NWR between 1985 – 1991. Fledging rates ranged from 0.04 to 2.1 chicks / nest (J. Dinan, pers. comm.).

Figure 39. Changes in number of breeding pairs at Goose Lake, Crescent Lake NWR, Nebraska, 1973 – 1996.

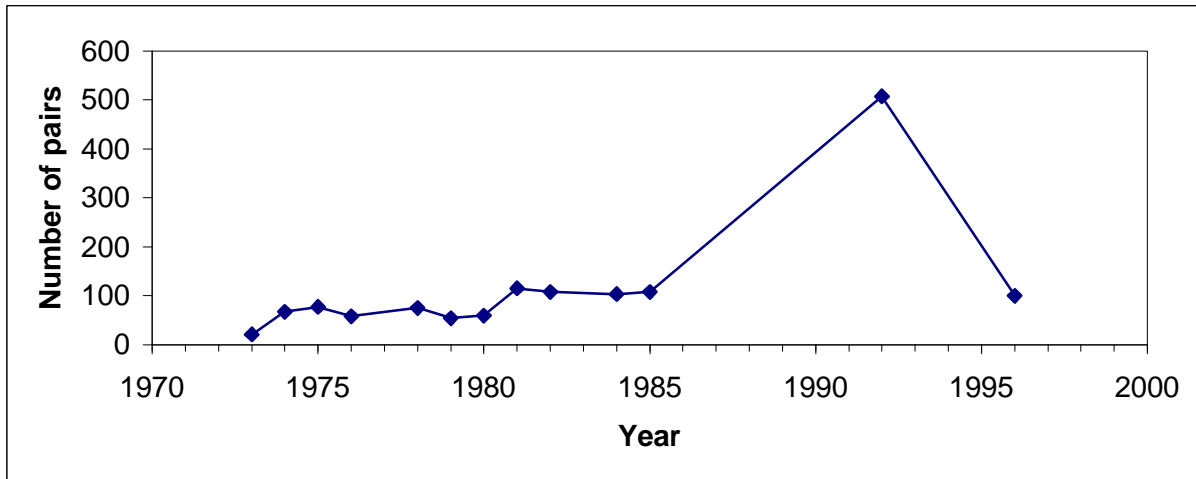
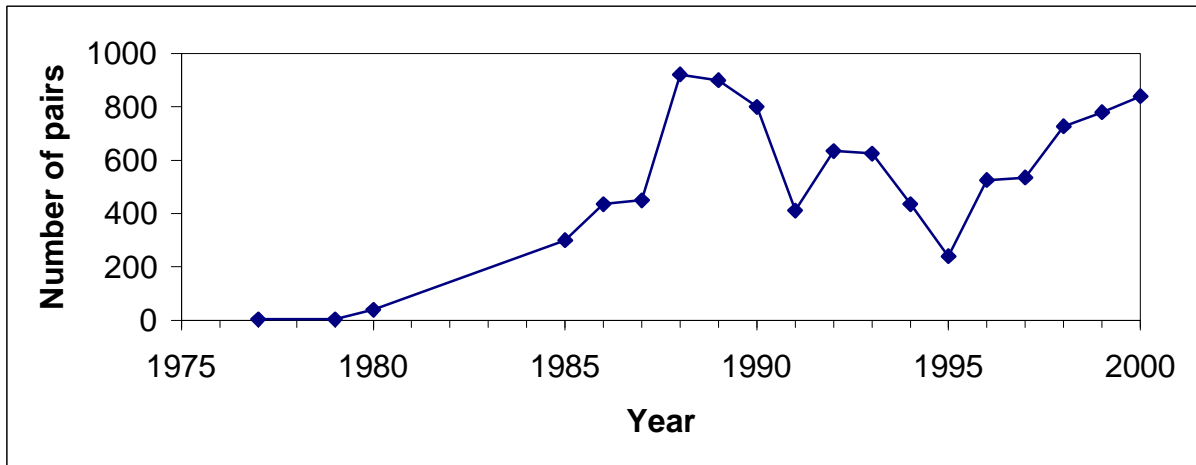


Figure 40. Changes in number of breeding pairs at Marsh Lakes, Valentine NWR, Nebraska, 1977 – 2000.





New York (Inland)

Summary of Population Data and Trends

(Survey completed by R. Miller, N.Y. State Dept. of Env. Cons., Delmar, NY; some breeding data from Weseloh and Ewins 1994.)

Breeding.

The Double-crested Cormorant was first reported breeding in inland New York in 1945. Over the last two and a half decades breeding numbers and locations have increased dramatically in this region. Nest counts at Little Galloo Island (the largest colony in the state, and one of the largest in North America) have been made yearly since 1974 (Weseloh and Ewins 1994; R. Miller, pers. comm.). This colony increased fairly slowly until the mid-1980s, then began to increase rapidly until 1996, when numbers peaked at 8,410 pairs. In 1998, numbers declined to 5,839 pairs (Figure 41). Between 1974 – 1992 numbers increased at an average annual rate of 35.8 %. However, the rate of increase has greatly slowed in recent years; between 1992 – 1998 the average annual rate of increase was only 1.2 %. As of 1997, there was a total of 12 known colony sites in upstate New York, with an estimated 9,072 pairs; most (84 %) were on Little Galloo Island [for entire state, upstate and Atlantic Coast, 12,346 pairs estimated at 21 colonies in 1997 – 1998, R. Miller pers. comm.; see New York (Coastal) profile].

Winter.

Cormorants do not winter in this part of the state (R. Miller, pers. comm.).

Migration.

During spring and fall migration, large flocks are observed along the eastern shore of Lake Ontario. Up to 2000 individuals have been reported at Oneida Lake in the fall (R. Miller, pers. comm.; Miller 1998). Fall migration peaks in late September; spring migration peaks late March – early April (R. Miller, pers. comm.).

Historical Information

The history of the Double-crested Cormorant and its distribution in New York was described by Miller (1997, 1998), and is summarized for upstate New York here, unless otherwise noted. In 1945, Kutz and Allen (1947) found a colony with about 14 nests on Gull Island in eastern Lake Ontario, about four miles from Henderson Harbor, Jefferson County. A long time resident reported that this was the first nesting of cormorants in the area in 75 years of his memory. This is believed to be the first authentic record of cormorants nesting in New York. The Gull Island colony supported about 20 pairs of cormorants and was presumed to be the only colony in the state for about two decades. As the Great Lakes cormorant population declined throughout the 1960s and early 1970s, the Gull Island colony probably died out in the 1960s.

Following the reduction of contaminants due to government anti-pollution programs and laws, cormorants made a remarkable recovery in the Great Lakes and elsewhere (see Zone 3: Summary of Population Data for Canadian and U.S. Interior Population Summary). In 1974, Little Galloo Island, which is near the former Gull Island colony, became occupied and began to increase steadily. New nesting sites at other upstate New York locations were later colonized, probably as a result of the expanding Great Lakes population, and colonies were established at Lake Champlain, Oneida Lake, and Lake Erie. In 1996, 8,410 pairs were counted on Little Galloo Island alone. In the summer of 1998, frustrated commercial fishing guides, believing cormorants responsible for lowered fishing success, illegally slaughtered 1500 – 2000 cormorants on Little Galloo. Ten individuals plead guilty to charges stemming from the slayings. Five who plead guilty to the most serious charges were sentenced to home confinement and fined. In spring of 1999, the USFWS issued a permit to New York state to oil eggs in up to 7,500 nests on Little Galloo Island to reduce recruitment into the Eastern Basin and to slow expansion of cormorants to new nesting sites. Control efforts are also undertaken on Oneida Lake, in the form of nest destruction during the breeding season, and harassment of migrant cormorants during the fall (VanDeValk et al. 1999; R. Miller, pers. comm.). See summary of New York (Coastal) for additional information on DCCOs in New York.

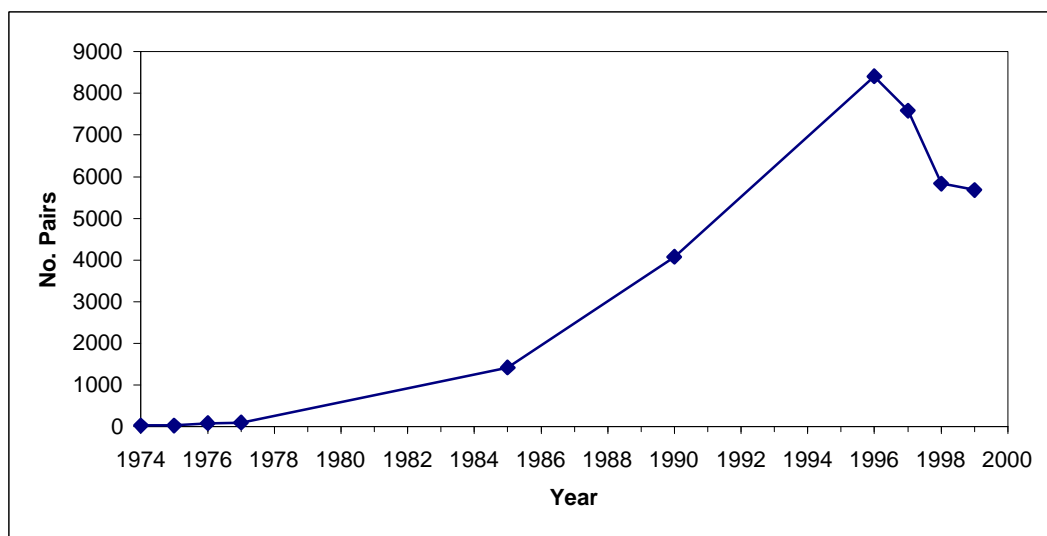
Land Ownership

Ownership information was provided for all 12 inland colonies. Seven were on privately owned lands, which were owned by NGOs such as The Nature Conservancy, private corporations, organizations and individuals. The other five were on public lands managed by federal and state agencies (R. Miller, pers. comm.).

Productivity

Weseloh and Ewins (1994) conducted undisturbed counts of medium-large young at the Little Galloo Island colony in July during 1981 – 1991, and estimated an average productivity of 2.14 young per active nest. On Wantry Island, Oneida Lake, 1.6 chicks per nest were allowed to complete incubation (VanDeValk et al. 1999).

Figure 41. Changes in number of breeding pairs on Little Galloo Island, 1977 – 1998.





North Dakota

Summary of Population Data and Trends

(Survey completed by C. Grondahl, North Dakota Game and Fish Dept.)

Breeding.

The DCCO has been documented as a nesting species in North Dakota since the late 1800s (see Lewis 1929 and Stewart 1975 for records). Currently, the DCCO nests in several areas across the state, but breeding surveys have not been conducted and precise locations are not known. Stewart (1975) reported “during recent years, the largest segment of the breeding population in the state has become established as scattered colonies along the margin of Lake Sakakawea or Garrison Reservoir. Other recent colonies have been found on the Oahe Reservoir about 10 miles below Bismarck and on scattered lakes and river impoundments located within the Prairie Pothole Region and in the Turtle Mountains.” He also described two distinct nesting habitats: all colonies along the Missouri River reservoirs nested in the tops of dead trees, mainly cottonwoods; colonies located on natural lakes were mostly on the ground on isolated islands. C. Grondahl (pers. comm.) suggested that, presently, all appropriate habitat east of the Missouri River system may have DCCOs. Some prime nesting locations include Devils Lake in the northeastern portion of the state, the back bays of upper Lake Sakakawea, several areas on Lake Oahe, and wetlands throughout central North Dakota. The southeastern and northeastern portions of the state also have plenty of good habitat. While no estimates of breeding numbers are available, personal observations of several North Dakota Game and Fish Department employees suggest that DCCOs have increased since 1975, when state status and distribution was reviewed by Stewart (1975) (C. Grondahl, pers. comm.).

Winter.

In winter, the DCCO does not occur (C. Grondahl, pers. comm.).

Migration.

No information was provided on migrants.

Historical Information

It is not clear when the DCCO was first recorded breeding in North Dakota, but some early observations of nesting colonies were made in 1897 at Sweetwater Lake, and in 1898 at Devils and Stump Lakes. The Devils Lake colony was active through 1920, and the Stump Lake colony was active as late as 1925. It is not clear when the Devils Lake colony was re-colonized. Some high counts prior to 1975 include 1,260 breeding pairs on all of Lake Sakakawea, 1958; 500 nests at Chase Lake, Stutsman County, 1967; 200 – 300 nests in one colony on Lake Sakakawea or Garrison Reservoir, 1970; 225 nests on the Upper Souris NWR, Renville County,

1965; 170 nests on J. Clark Salyer NWR, McHenry County 1946 (see Stewart 1975 for these and additional records).

Land Ownership

No information provided.

Productivity

No information provided.



Ohio

Summary of Population Data and Trends

(Survey completed by M. Shieldcastle, Crane Creek Wildlife Research Station, Oak Harbor, OH.)

Breeding.

In 1987, the Double-crested Cormorant returned to Ohio as a nesting species, and though no chicks were fledged from this attempt, this nesting provided the first nesting record for the state in this century (see Historical Information, below). The first successful nesting was documented in 1992 at West Sister Island, where numbers increased steadily through 1995. A very large increase occurred between 1994 – 1995, then for the next 3 years remained fairly stable (Figure 42). In 1998, 1500 pairs were estimated at West Sister Island. Overall, this colony increased at an average annual rate of 41.6 % since discovery. A second small colony of 10 pairs was found at Mercer Wildlife Area in 1998, bringing the state total to 1510 pairs (M. Shieldcastle, pers. comm.).

Winter.

A small number of birds (50) are estimated to winter in the state (M. Shieldcastle, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 individuals were estimated during any count, but small numbers were observed at some sites.

Migration.

Large numbers (25,000) are reported during peak migration, which occurs in the spring approximately April – May, and in the fall August – October. Migrants are observed roosting on islands around and feeding on western Lake Erie, including Sandusky Bay (M. Shieldcastle, pers. comm.).

Historical Information

In the early 1800s cormorants were regular migrants along Lake Erie. The creation of canal reservoirs in the state provided new habitat, breeding populations formed in the 1860s and 1870s at Buckeye Lake and Lake St. Mary's (Peterjohn and Zimmerman 1989). However, cormorants may have been nesting at Buckeye Lake earlier than this, since Wheaton (1882) writes that the species "was said to have nested years ago at the Licking Reservoir [Buckeye Lake]." There is also some confusion over how many colony sites were actually documented. Lewis (1929) reported that prior to the 1900s, cormorants nested at Grand Reservoir, St. Mary's Reservoir and possibly at Licking County. However, Grand Reservoir and St. Mary's Reservoir may be the same site, since they both occur in the same counties (Mercer and Auglaize), and today Lake St. Mary's is called Grand Lake St. Mary's. "Large numbers" of birds were reported to nest at St. Mary's colony, while only 10 – 15 pairs were recorded at the Buckeye Lake site. However, these colonies were subject to indiscriminant hunting and egg collecting (Peterjohn and Zimmerman 1989). Numbers decreased rapidly at St. Mary's Reservoir after 1867 (Langdon 1878), and by the early 1880s, this colony and the one at Buckeye Lake had disappeared (Henninger 1904; Peterjohn and Zimmerman 1989). By the turn of the century (or earlier) the cormorant no longer nested in the state (Dawson 1903; Jones 1903).

Between 1900 and 1950 migrants were still observed during the spring and fall along Lake Erie, where they were uncommon to fairly common, though flocks were seldom larger than 10 – 20 individuals. In the late 1950s, reduced numbers became evident, and by 1965 they were rare throughout Ohio with < 10 sightings annually of 1 – 5 individuals. Declines in the state were thought to have resulted from widespread DDT use and other contaminants throughout the region (Peterjohn and Zimmerman 1989).

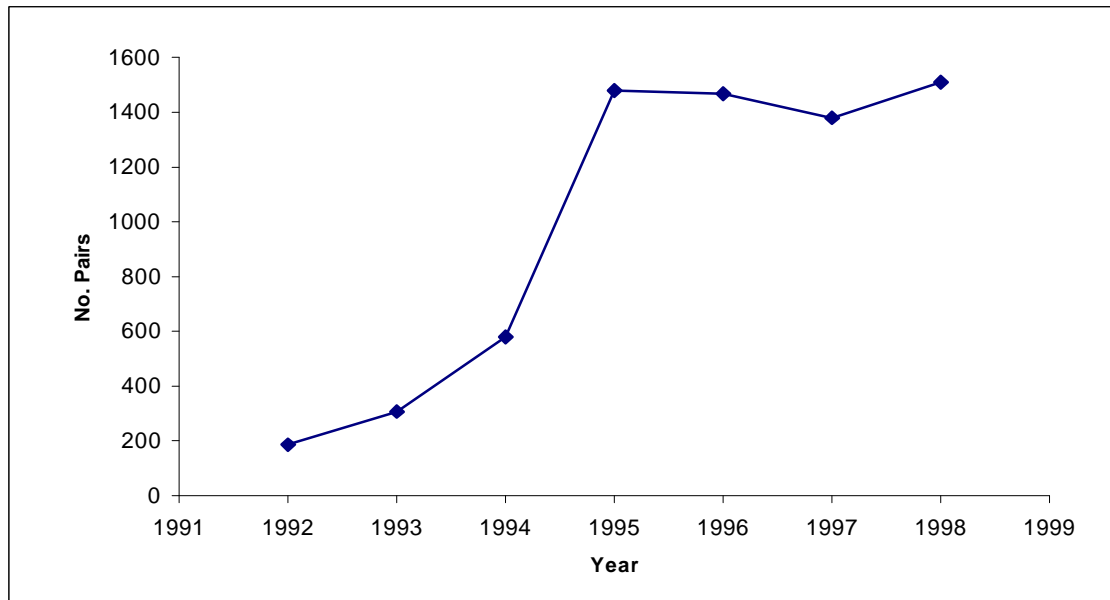
In the 1980s, the cormorant population began to recover in Ohio and hundreds of birds were recorded during migration. Large concentrations (500 – 2000) along western Lake Erie, a major staging area for the Great Lakes nesting population, have become a regular fall occurrence. In 1987, six pairs nested at the Ottawa Wildlife Refuge on w. Lake Erie, and though these nests ultimately failed, this provided Ohio's first nesting record in this century.

Land Ownership

Land ownership was provided for both breeding sites. The West Sister Island colony is on federal land, owned by the USFWS; the Mercer WA is on state land, owned by the Ohio Division of Wildlife (M. Shieldcastle, pers. comm.).

Productivity

No estimates available.

Figure 42. Changes in numbers of breeding pairs in Ohio, 1992 – 1998.

Ontario

Summary of Population Data and Trends

(Survey completed by J. Harcus, Ministry of Natural Resources, Petersborough, Ontario; breeding data provided by D.V. Weseloh, Canadian Wildlife Service, Downsview, Ontario.)

Breeding.

Ontario has the largest number of breeding pairs of Double-crested Cormorants in the Great Lakes region, and perhaps the second largest on the continent. Between 1980 – 1997, there was a total of 116 known colony sites; in 1997, 74 colonies were active, and there was an estimated 35,159 pairs nesting in the St. Lawrence River and on Lakes Ontario, Erie, Huron and Superior. Large increases have occurred over the last 25 years (Figure 43); these have been attributed to decreased contaminant loads in the Great Lakes, and to significant increases in alewife and rainbow smelt (Weseloh and Collier, 1995; Weseloh et al. 1995). However, growth appears to be slowing in several locations, e.g. the North Channel, Lake Huron; Lake Ontario; and Lake Superior. Between 1981 – 1990, the average annual rate of increase was approximately 38 %; but D.V. Weseloh reported that between 1993 / 1994 – 1997, the annual rate of increase was only 3.7 % (*in* Korfanty et al 1997). Numbers in the upper Great Lakes are expected to level off in the next decade (Ludwig and Summer 1997).

Winter.

Birds are not known to winter in the province (J. Harcus, pers. comm.).

Migration.

Migrants do not pass through, as birds found in the province originate there (J. Harcus, pers. comm.).

Historical Information

The history of the Double-crested Cormorant in Ontario was described by Korfanty et al. (1997) and is summarized here, unless otherwise noted. Cormorants have nested in Ontario for hundreds of years; the first documented breeding dates back to 1758, when cormorants were recorded breeding at Lake of the Woods, where they continue to nest today. Between 1900 – 1920 they moved eastward to lakes Superior and Nipigon; the first documented nesting in the Great Lakes occurred on the far western end of Lake Superior in 1913. Lewis (1929) reported active colonies in eastern L. Superior, Lake Nipigon and Lake of the Woods between 1927 – 1928 with an estimated 80 pairs. In 1931 they began nesting in the North Channel of Lake Huron, and by 1938 and 1939, breeding colonies were established on lakes Ontario and Erie, respectively. By the 1950s, the cormorant population on the Canadian Great Lakes peaked at about 900 nests, an abundance that led sport and commercial fishermen to call for a cormorant control program. In response, a cormorant control program was introduced in Ontario, primarily on Georgian Bay. Efforts included destruction and spraying of eggs. Illegal control also occurred, with fish harvesters shooting adults, and destroying eggs, nests and young. However, cormorant control only slowed the growth of the cormorant population in the 1940s and 1950s, control was continued on the Canadian Great Lakes until 1966.

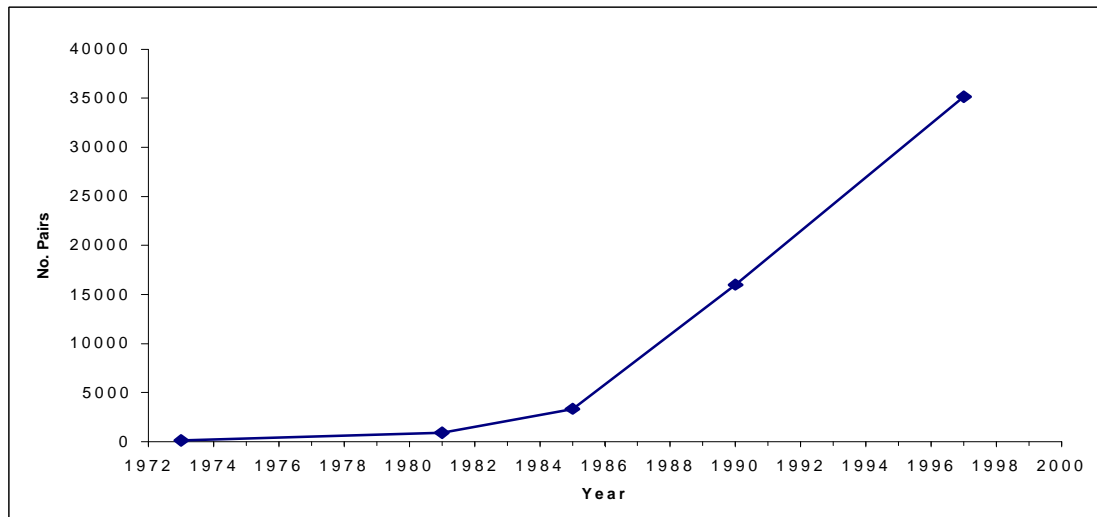
Between the 1950s and the 1970s, the number of nesting pairs on the Canadian Great Lakes was reduced by 86 %. This reduction is thought to have occurred mainly as a result of toxic chemical concentrations (e.g. DDT, DDE, and PCBs) in the Great Lakes and their impact on cormorant reproduction. DDT and other pesticides were banned in the 1970s, and as levels of contaminants declined in the Great Lakes, cormorant reproduction began to improve and cormorant numbers began to dramatically increase.

Land Ownership

No land ownership information was provided.

Productivity

No productivity information was provided.

Figure 43. Changes in breeding numbers in Ontario, 1973 – 1997.

Saskatchewan

Summary of Population data and Trends

(Survey completed by K. Roney, Royal Saskatchewan Museum, Regina, Saskatchewan.)

Breeding.

In Saskatchewan, there has been a total of 19 known colonies since 1968, but some have become inactive. In 1980, there was a total of 14 known colonies; in 1991, the last year Double-crested Cormorants were censused, there were 10 active colony sites with an estimated 19,547 pairs (K. Roney, pers. comm.). Complete censuses have been conducted since 1968, at which time 1078 pairs were estimated (Vermeer 1970). Numbers appear to have steadily increased (Figure 44); the average annual growth rate between 1968 – 1991 was 13.4 %. The greatest growth occurred during the 1970s; between 1985 – 1991, growth slowed to 2.7 %. The species is currently protected under the province's Wildlife Act (K. Roney, pers. comm.), but we were unable to determine the exact date when the species received province-wide protection (see Historical Information, below).

Winter.

No information provided.

Migration.

No information provided.

Historical Information

The earliest mention that we found of the Double-crested Cormorant in the Saskatchewan area was by Nuttall (1834, cited in Mendall 1936), who noted that a DCCO (identified as *P. dilophus*) was collected on the Saskatchewan River in the month of May. Audubon (1843) also noted that the species bred on the Saskatchewan River. The species has been nesting in the province for over a century. W. Spreadborough found a colony with 27 nests at Crane Lake in 1894, and J. Macoun reported breeding at Old Wives Lake in 1895; around this time, the species was reported as “abundant and breeding westward to Old Wives Lake and Crane Lake,” and to breed in all suitable places around Prince Albert and at Big Stick Lake (Macoun and Macoun 1909, cited in Lewis 1929). Seton (1908) reported a large colony at Lake Isle a la Crosse. By the time Lewis was writing, breeding appears to have ceased at Old Wives, Crane and Big Stick lakes, but was documented at 9 other locations with roughly 1600 pairs.

During 1950 – 1965, large scale destruction of nests by fishermen and ranchers occurred in the province because cormorants were perceived as a threat to fishing interests. The first thorough survey conducted in 1968 found nine colonies, though one or two may have been missed. Several colonies had disappeared by this time due to declining lake levels and human disturbance, and the breeding population had declined since the late 1920s (Vermeer 1970). In 1971, five colonies were given protection, and numbers have increased dramatically in the province since that time (Vermeer and Rankin 1984; K. Roney, pers. comm.).

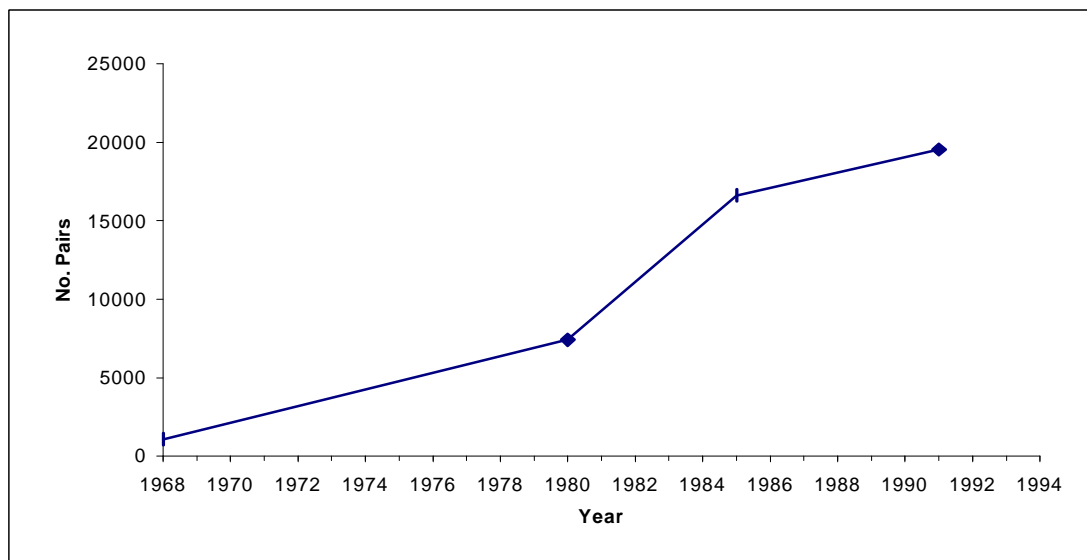
Land Ownership

Ownership information was provided for eight of the most recently active colonies. Most were on public lands in either wildlife refuges, bird sanctuaries, or National Park lands; one was on land that is part of a National Defense Area (K. Roney, pers. comm.).

Productivity

No estimates provided.

Figure 44. Changes in breeding numbers in Saskatchewan, 1968 – 1991.





South Dakota

Summary of Population Data and Trends

(Survey completed by E. Dowd-Stukel and D. Backlund, South Dakota Dept. of Game, Fish and Parks; breeding data supplied by R. Peterson, South Dakota Ornithologist's Union.)

Breeding.

The DCCO has been documented as a nesting species in South Dakota since at least the late 1800s (see Historical Information, below). Between 1988 – 1992, surveys were conducted during the South Dakota Breeding Bird Atlas, and 36 DCCO colonies were documented (Peterson 1995). Of these, 10 were on human influenced habitats (e.g., flooded timber of the Missouri River reservoirs and other impoundments, E. Dowd-Stukel, pers. comm.), while 25 were on natural habitat; habitat type for one colony was not reported. The majority of colonies are found in the eastern portion of the state, and nesting habitat includes islands, sandbars and trees near lakes (Whitney et al. 1978). Colonies also occur along the Missouri River. In the Sand Lake NWR all nests were in cattail marshes on floating mats. Large numbers occurred in Kingsbury County, where a total of 2,962 pairs nested in 11 colonies in 1991 (Peterson 1995). No data post-1992 were available. Because regular statewide censuses are not conducted, information on breeding trends and state breeding total is not available. Colonies along the Missouri River may be in decline as flooded timber disappears. Formerly large colonies at Crow Creek and Grand River are much smaller, and this is probably true on most reservoirs (E. Dowd-Stukel and D. Backlund, pers. comm.).

Winter.

In winter, not reported to occur (E. Dowd-Stukel and D. Backlund, pers. comm.). The South Dakota Ornithologists' Union (1991) notes that early winter records are probably late or crippled migrants.

Migration.

No areas known to provide important stopover sites for migrants (E. Dowd-Stukel and D. Backlund, pers. comm.). In spring, migrants are normally observed the first two weeks of April; in fall, mid-September to early October (Whitney et al. 1978; South Dakota Ornithologists' Union, 1991).

Historical Information

The earliest record we obtained was that of Agersborg (1885, cited in Lewis 1929), who reported that "DCCOs once nested commonly in trees along streams in southeastern South Dakota, but that with the cutting of the timber they abandoned the region." Lewis (1929) reported "present colonies" only in the northeastern portion of the state. No additional information was obtained.

Land Ownership

No information provided.

Productivity

No information provided.

**Utah****Summary of Population Data and Trends**

(Survey completed by F. Howe, Utah Division of Wildlife Resources.)

Breeding.

The DCCO has been documented as a nesting species in Utah for the last 150 years (see Historical Information, below). The subspecies occurring here is believed to be *P.a. auritus* (Hayward et al. 1976). Between 1971 and 1999, a total of eight colonies was known to be active. Consistent data are not collected on DCCOs in this state, so statewide totals and precise population trends are not available. However, up until 2000, Utah's largest colony and most of Utah's breeding population occurred at Bear River Refuge, where data have been collected regularly. Numbers at Bear River have fluctuated since 1971, but increased overall up to 1997. In 1998 and 1999, declines occurred, and in 2000 the unit was dry due to construction and no nesting occurred (Figure 45). With the exception of Great Salt Lake (GSL), all other colonies have been small (≤ 55 pairs). At GSL, no information has been obtained on numbers of breeding pairs or nests, though numbers of birds were counted each July at five state Waterfowl Management Areas on the east shore of GSL between 1980 – 1992. During these counts, the number of individual birds ranged from 29 to 429 (F. Howe, pers. comm.).

Winter.

In winter, the DCCO is not reported to occur (F. Howe, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 individuals were estimated during any count, but small numbers were observed at a few sites.

Migration.

Peak fall migration occurs mid-to-late September; no information was provided for spring migration. The Bear River Refuge was reported as a significant stopover site for migrant birds, with numbers ranging from 2 – 500 birds (F. Howe, pers comm.).

Historical Information

Stansbury (1852) gave the first description of DCCO colonies on Egg Island, Great Salt Lake, when he visited this site in 1850. Ridgway (1877) also recorded the species around Great Salt Lake. There are numerous early nesting records from this area, and early breeding colonies

reported in the literature all occurred in northern and western Utah. Records obtained between 1926 – 1942 include nesting at Egg Island, Great Salt Lake, in Salt Lake County, and in Cache County, five miles west of Logan. The latter colony nested in dead trees along the banks of Little Bear River (Bee and Hutchings 1942). Breeding colonies have also been reported in Rich, Millard, Box Elder, Davis and Utah counties. Between the 1920s and 1970s numbers of nesting DCCOs decreased in Utah. In 1974, a total of 13 colonies was known to have existed in the state, but only five were still in use. During the 1930s and 1940s the larger colonies may have supported a combined total of > 1000 birds, but by 1973 a total of 386 birds (number of birds, not pairs, reported) was estimated to nest in five colonies (Mitchell 1977).

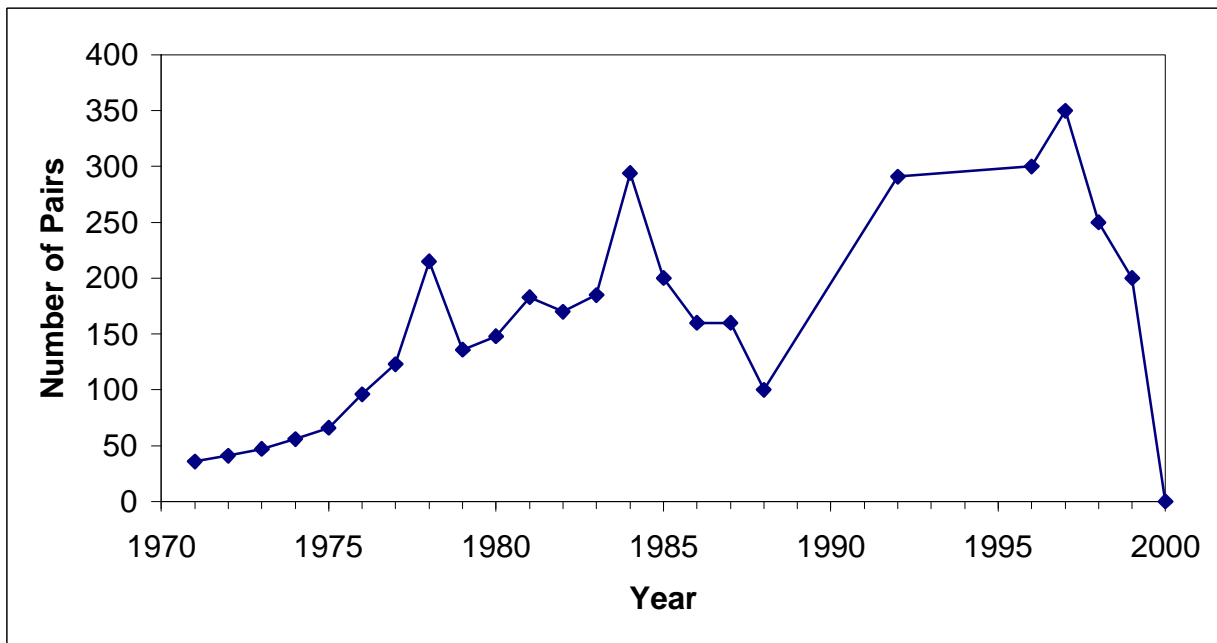
Land Ownership

Information was provided only for the Bear River Refuge, which is federal land managed by USFWS (F. Howe, pers. comm.).

Productivity

No information provided.

Figure 45. Changes in breeding numbers at Bear River Refuge, Utah, 1971 – 1996.



Vermont

Summary of Population Data and Trends

(Survey completed by L. Garland, Vermont Fish and Wildlife, Essex Junction, VT.)

Breeding.

The Double-crested Cormorant appears to be a fairly recent breeder in Vermont, with the first known nesting documented in 1982 (Laughlin and Kibbe 1985). In 1999, there were five known colony sites; two were active with a total of 2805 pairs. After initial discovery numbers slowly increased for the first seven years. By 1995, numbers had nearly quadrupled since 1989, and have continued to steadily increase (Figure 46). Over the last decade numbers increased at an average annual rate of 22 %. Nearly all of the growth has occurred on Young Island; in 1999, > 90 % of pairs were on this island (L. Garland, pers. comm.).

Winter.

No significant numbers winter in the state (L. Garland, pers. comm.).

Migration.

No significant concentrations are reported during migration (L. Garland, pers. comm.).

Historical Information

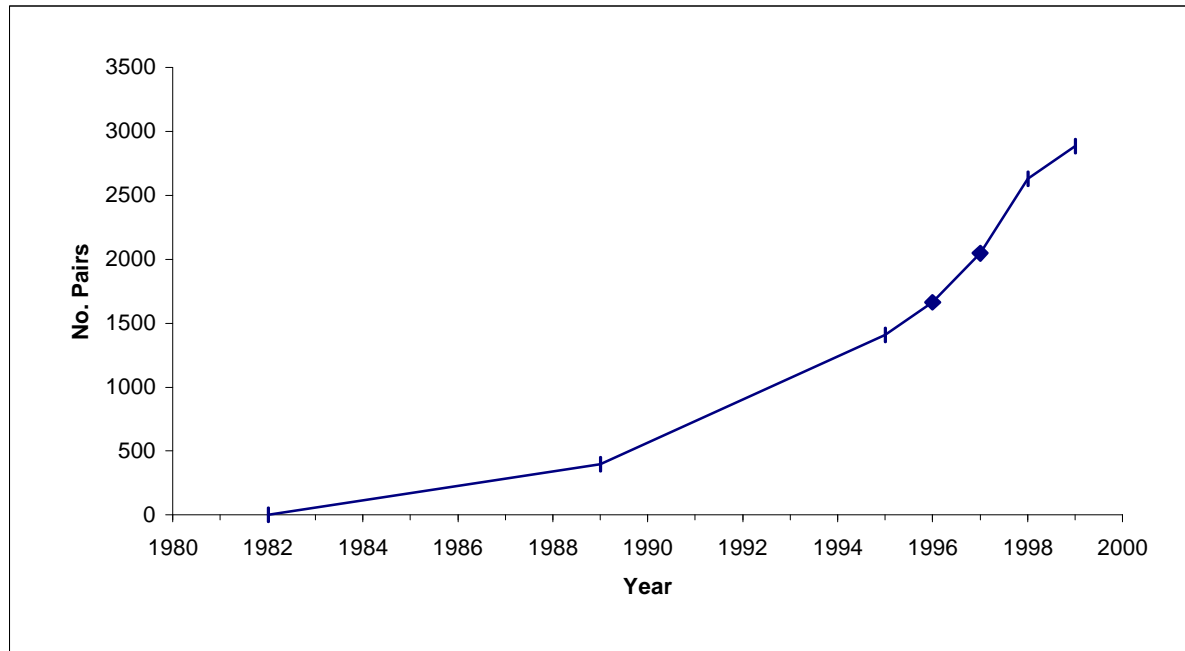
The earliest records we were able to obtain were from the 1930s, when three specimens and one sight record were listed for the state. During the Vermont Atlas Project Survey period (1976 – 1981) observers documented increasingly frequent summer occurrences of Double-crested Cormorants on Lake Champlain, especially around Young Island. The first nesting for the state was documented on Young Island the year after the atlas survey, 1982; one nest was observed in a tree. Breeding was not recorded at other sites until 1994, when cormorants expanded nesting to Mud Island. In 1996 they expanded further to Bixby, Shad and Popasquash Islands. Between 1994 – 1997, nests were removed from Mud, Bixby and Popasquash Islands. In 1998, cormorants did not return to nest at Mud or Popasquash Islands, but did return to Bixby Island, where harassment techniques prevented successful nesting and eventually drove the birds from the island (L. Garland, pers. comm.).

Land Ownership

Both Young and Mud Islands are on state land; Shad Island is on federal land; Popasquash is on private land, owned by the Green Mountain Audubon Society; and Bixby Island is owned by a private individual (L. Garland, pers. comm.).

Productivity

Fowle et al. (1997) estimated productivity on Young Island on Lake Champlain in 1995 and 1996. Multiplying the mean number of young per nest by the number of nests for each season, they estimated that 2.54 young fledged per nest in 1995 and 1996.

Figure 46. Changes in numbers of breeding pairs in Vermont, 1982 – 1999.

Wisconsin

Summary of Population Data and Trends

(Survey completed by S. Matteson, Wisconsin DNR, Bureau of Endangered Resources, Madison, WI.)

Breeding.

Wisconsin has the second largest number of breeding pairs in the U.S. Great Lakes. In 1997, there was a total of 46 known colony sites. Of these 23 were active, with a total of 10,546 pairs. Seventeen colonies are on the Great Lakes (Apostle Islands, Lake Superior; Green Bay / Lake Michigan Islands); the rest are on inland lakes, marshes and the Upper Mississippi River. However, the Great Lakes colonies, particularly those on the Green Bay / Lake Michigan Islands, support a far greater number of nesting pairs than do the inland colonies. By 1990, the number breeding on the Great Lakes was about five times larger than that utilizing inland habitat. Increases in breeding numbers have occurred steadily since 1973 (Figure 47). Matteson et al. (1999) used a regression model to calculate trends in the state during 1973 – 1997, and reported an average annual increase of 24.7 %; they also noted that the rate of increase significantly slowed in recent years, 13.2 % during 1986 – 1997, compared to 36.9 % during 1973 – 1985. During the more recent period, increases were significantly greater on Great Lakes sites than on inland sites.

Winter.

Cormorants are not known to winter in the state in any significant numbers (S. Matteson, pers. comm.).

Migrants.

Spring migration occurs late March – late April; fall migration, mid-to-late August – November. During fall migration several thousand (5,000 – 7,000 in 1991 and 1992) are seen on the Upper Mississippi River, and large numbers congregate around LaCrosse and Trempeleau (Kirsch 1995).

Historical Information

Around 1875 DCCOs were common migrants in suitable waters throughout Wisconsin. But by the turn of the century, numbers of migrants appear to have declined; Kumlien and Hollister (1903) note that “even when it was more common, comparatively few were noticed in the interior during the fall.” However, large numbers were still at least occasionally observed into the 1920s; on April 24, 1926, a huge number of birds was observed migrating up the Mississippi River past LaCrosse, estimated at 100,000 to 1,000,000 individuals, a flock size which by this time was considered “unusual” for the region. The flock was so large “that at times it was impossible to see the sunset sky through the mass” (Grassett 1926). USFWS unpublished reports note that 20,000 to 40,000 cormorants used the Upper Mississippi River during spring and fall migration between 1939 – 1950 (Kirsch 1995). Anderson and Hamerstrom (1967) also report that “large” numbers of migrants, 2,000 – 5,000 individuals, were still reported in the 1940s and 1950s, but these are nothing like the huge numbers reported by Grassett (1926). By the late 1950s, migrants appeared to decline and continued to do so through the 1960s, with none to very few reported during the migration (Anderson and Hamerstrom 1967). Numbers of migrants on the Upper Mississippi River have apparently increased some since the late 1960s, but have remained much lower than those reported in the 1940s (Kirsch 1995).

The breeding history of the cormorant and changes in its numbers and distribution in Wisconsin were reviewed thoroughly by Matteson et al. (1999), and are summarized here. The first published reports of nesting did not occur until 1919 and 1921, when cormorants nested on Lake Wisconsin (formerly Okee Flowage) in south-central Wisconsin. However, Kumlien and Hollister (1903) report communications from C.F. Carr that cormorants nested earlier than this on “some of the larger, isolated lakes in the northern and central part of the state.” Additionally, Anderson and Hamerstrom (1967) report that cormorants probably bred in most of the state in the early 1900s. From 1921 to the mid-1950s breeding records were regularly obtained; during this time cormorants occupied 17 known colony sites in 16 counties. The total number of nesting pairs reached at least several hundred in peak years. However, from the 1950s to the early 1970s, numbers of breeding cormorants and colony sites sharply declined due to combined effects from bioaccumulation of DDT and its metabolites, human persecution and habitat loss. By 1966 only three active colony sites (24 pairs total) were identified; in 1972 the Double-crested Cormorant was officially listed by the state as endangered.

The installation of artificial nesting platforms, declines in DDE levels and protection as an endangered species led to a marked recovery. In 1973 a statewide survey revealed that the small population in 1966 had more than doubled, with a total of 66 pairs counted. By 1982 the state population had increased to 1028 pairs in 16 colonies, and due to its comeback, its official status was changed from “endangered” to “threatened.”

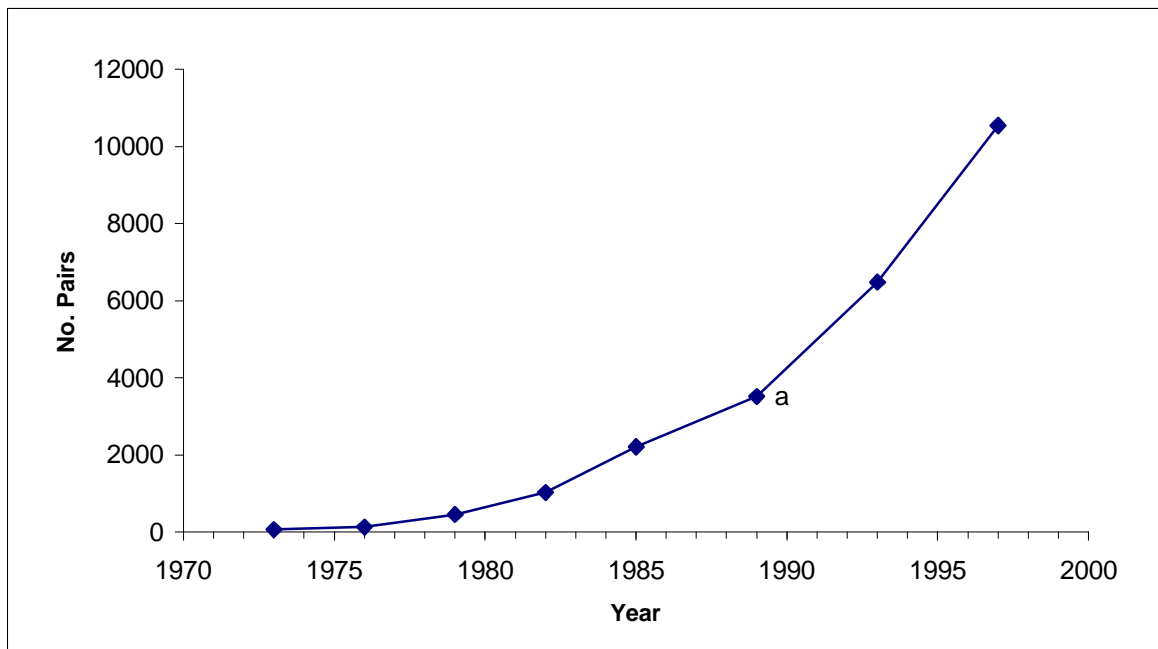
Land Ownership

No information provided.

Productivity

Statewide, mean annual production per nesting pair during 1973 – 1997 was 1.58, with a range of 0.00 – 3.36 young per nesting pair (Matteson et al.1999).

Figure 47. Changes in breeding numbers in Wisconsin, 1973 – 1997.



a = conservative estimate, as not all colonies were censused in this year (Matteson et al. 1999)



Wyoming

Summary of Population Data and Trends

(Survey completed by A. Cerovski, Wyoming Game and Fish Dept.)

Breeding.

The DCCO has been documented as a breeding species in Wyoming since 1928 (see Historical Information, below). Since 1981, a total of 25 colonies have been documented, and occur throughout the state, with the exception of the southwest (Figure 28). All known colonies are not censused yearly, and precise information on trends is not available, nor is an estimate of the total number of birds breeding in the state. However, Findholdt (1985) reported a dramatic increase in numbers of nesting DCCOs between historical times and the mid-1980s. The main

reason for this increase is thought to be construction of reservoirs with isolated islands. In addition to supplying breeding habitat, reservoirs likely increased prey abundance and availability. Also, decline in contaminants may also have enabled numbers to increase.

Between 1986 – 1994, numbers appear to have declined. In 1986, 21 colonies were counted; of these, 16 were active and a total of 1,457 nests was estimated. Large numbers (113 – 303 pairs) occurred at Cooper and Ocean Lakes, Pathfinder Reservoir Bird Island, Soda Lake West Island and Soda Lake Manmade Island. In 1994, 15 colonies were censused and an additional two were known to be active. At most colonies declines occurred. At the censused colonies, a total of 364 nests was estimated, and only one large (≥ 100 pairs), located at Yellowstone Lake Molly Island (125 pairs), was reported. Eight colonies were known to be inactive. Changes in water levels were responsible for lack of breeding at least one colony, Cooper Lake (A. Cervoski, pers. comm.).

Winter.

In winter the DCCO is not known to occur in any significant numbers (A. Cervoski, pers. comm.).

Migration.

There are no known important migrant stopover sites, and no estimates of migrants were provided (A. Cervoski, pers. comm.).

Historical Information

The earliest records of DCCOs in Wyoming that we obtained were reported in Grave and Walker (1913) who note there are “but three Wyoming records, two from the Laramie Plains and one from near Buffalo.” The first confirmed breeding record was obtained on Molly Islands, Yellowstone Lake, at Yellowstone National Park in July 1928 (see Findholdt 1985 for records). Observations of small nesting numbers (≤ 25 pairs) were occasionally obtained at this location until the mid 1980s, at which time numbers began to increase. In the late 1980s – mid 1990s, 85 – 148 pairs nested in any one year. Additional colonies were documented on the North Fork Arm of the Shoshone Reservoir in 1938, at Ocean Lake in 1942, and at Bamforth Lake in 1972 (see Findholdt 1985 for records). In 1984, Findholdt (1985) documented 15 active and five inactive colonies.

Land Ownership

Land ownership information was supplied for 20 colony sites. Of these, nine were on public lands, while the other 11 were owned by private individuals or corporations (A. Cervoski, pers. comm.).

Productivity

Productivity information (number of chicks fledged / nest) was provided for the colony located at Yellowstone Lake Molly Island. Between 1985 – 1996, productivity ranged from 1.4 to 2.2. Productivity rates at this colony showed nearly steady increases during this time period, and the colony has grown fairly steadily from 25 nests in 1981 to 126 nests in 1996 (A. Cervoski, pers. comm.).



ZONE 4: SUMMARY OF POPULATION DATA FOR SOUTHEASTERN U.S.

Introduction

Zone Boundaries.

In addition to Florida, Hatch and Weseloh (1999) included Georgia, Alabama, Mississippi and Louisiana in the southeastern breeding zone, but noted that birds are generally rare or absent as breeders from these states. However, as numbers grow in the southeast, this description may no longer hold true. With the exception of Alabama, breeding is currently occurring in small numbers in all of these states. Additionally, breeding is occurring in other southern states, such as North and South Carolina, Tennessee, Arkansas, Oklahoma and Texas. Some of these nestings are recent, while some represent re-colorizations. Based mainly on winter distribution of cormorants and possible *floridanus* subspecies designations (birds in TX, NC and SC, see below), we have expanded the boundaries of this population zone to include all of these southern states.

As noted in the introduction to this section, the winter range is a succinct geographic unit and it would not make biological sense to split it up. Additionally, birds breeding in some southern states may be year-round residents, while birds from the interior and northeast Atlantic are migratory. Another factor that influenced our decision to include additional states in this zone is that birds breeding in Texas may be the *floridanus* subspecies (Palmer 1962; Johnsgard 1993), though it is possible that birds breeding in northwest Texas may be the *auritus* or *albociliatus* subspecies, and be part of the interior or Pacific Coast populations, respectively. Similarly birds breeding in Oklahoma could be part of the interior Zone 3 population (Palmer 1962). Hatch and Weseloh (1999) included birds breeding in the Carolinas with the northeast Atlantic population, but the possible *floridanus* subspecies designation of these birds (Palmer 1962; Clapp and Buckley 1984; Post and Post 1988), along with winter distribution, prompted us to include them with the southeast population.

Breeding Information

With the exception of Florida, numbers of Double-crested Cormorants breeding in this zone are marginal; compared to Zones 3 and 5, numbers have remained small over the last two decades. The species has just recently returned as a breeder to Arkansas, Mississippi, and Tennessee, and is thought to have just recently colonized South Carolina and Georgia. These latter states were the only states in this zone that had enough consistent data to estimate trends in breeding numbers; in both of these states numbers are increasing, though total numbers are insignificant compared to increases in Zones 3 and 5. If the estimate of breeding pairs in Florida is fairly accurate (see Florida summary), total number of breeding pairs in this zone is probably < 10,000 (Table 5).

Winter and Migrant Birds

The southern and southeastern coastal U.S. provides much of the wintering habitat for Double-crested Cormorants breeding in the interior and along the northeast Atlantic Coast of the continent (Zones 3 and 5). Of the zones where DCCOs winter, this one provides habitat for the greatest number of birds (however, numbers wintering in Mexico are unknown). Most of the birds that breed in this zone are year round residents, based on the Florida population alone. Over the last few decades, numbers of wintering cormorants appear to have increased dramatically in

this zone [based on surveys conducted by Wildlife Services and CBC data (Sauer et al. 1996)]. Data from CBCs conducted between 1959 – 1988 have been analyzed for several states in this zone, and show increases ranging from 3.5 – 18.7 %. The largest increases occurred in Louisiana, Mississippi, Oklahoma and Texas. Data collected during the last 10 years suggest that numbers are still increasing in these states and others, but these data have not yet been analyzed.

Based on CBCs conducted between 1984 – 1997 (Sauer et al. 1996), the number of birds occurring in December in southeastern U.S. ranged between 105,000 – 313,000 (Figure 48). This is an obvious underestimate, given that our incomplete estimate of breeding individuals from the interior population alone is > 300,000 birds, and does not include young of the year, non-breeders or breeders from several states and provinces. Because CBC sites are selected to maximize variety of species and numbers counted, they do not sample all areas where cormorants are likely to occur, such as important roost sites around aquaculture facilities. Therefore, they tend to underestimate numbers (see data for Mississippi and Georgia, Table 5, and state summaries); however, they do provide useful information on trends and are used here as a general index of population change (see Intro for this section). Figure 51 compares distribution of DCCOs during winter months based on CBC data and winter roost locations obtained during surveys conducted by USDA/WS in Alabama, Arkansas, and Mississippi.

Historical Information

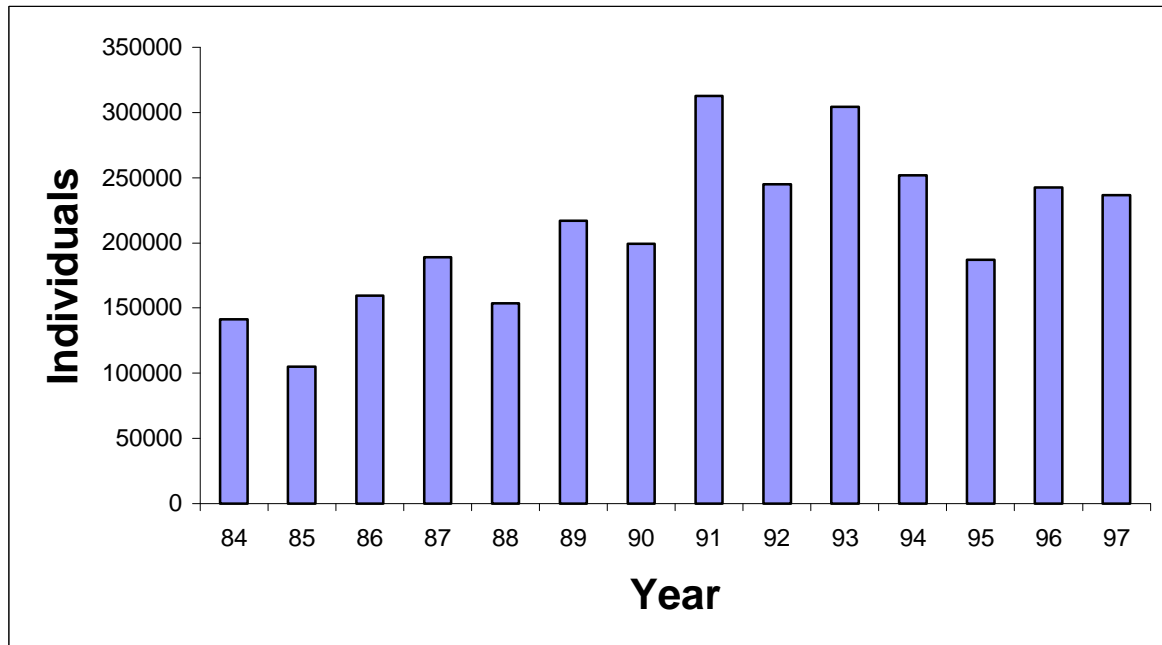
Prior to the twentieth century, the Double-crested Cormorant appears to have been an abundant year round resident, both as a breeding and wintering bird, along the Gulf Coast from Texas to Florida, and along the southern Atlantic Coast up through the Carolinas. Audubon (1843) described the *floridanus* subspecies as “constantly resident in the Floridas and their Keys, and along the coast to Texas.” He noted that the *auritus* subspecies wintered on the eastern coast, but rarely farther south than the Capes of North Carolina, where it encountered *floridanus*. Breeding was documented in the Carolinas, at least North Carolina, as early as the 18th century. Though breeding was not documented in most of the southern states until the 20th century, based on Audubon’s (1843) and other early observers’ comments, breeding may have occurred earlier in states such as Arkansas, Kentucky, Tennessee, Mississippi and Texas (see state summaries).

Information on status around the turn of the century is very limited; however, declines in breeding numbers in Arkansas and North Carolina were apparent by this time. In other states, such as Tennessee and Louisiana, declines in breeding numbers occurred later, between the 1940s and 1970s; also declines occurred again in North Carolina during this time. In Mississippi, there is little information on breeding, but numbers reported in the 1950s appeared to be in decline. Based on Christmas Bird Count data, significant declines in wintering numbers in the mid-south states also occurred during this period. Declines have been attributed to human persecution, habitat degradation and pesticides (Howell 1911; Jackson and Jackson 1995).

During this low point in cormorant numbers, the catfish industry in the mid-South underwent its major growth. Commercial catfish farming got underway in Kansas and Arkansas in the 1950s. By 1965, the first commercial catfish pond was established in Mississippi, and over 7,000 acres were developed for aquaculture in Arkansas. Texas and Louisiana also had significant acreage. However, Mississippi quickly became the industry leader, and by 1985 had over 85,000 acres of commercial catfish ponds (Stickley and Andrews 1989). As breeding populations in the interior and along the Atlantic Coast began to recover in the mid-to-late 1970s, numbers of wintering cormorants began increasing in the South. Increases in the Mississippi Delta and Texas have been especially dramatic.

Jackson and Jackson (1995) point out that the timing of the mid-south breeding population decline and recent increase coincides with the DDT era, the subsequent banning of DDT and the decline of pesticide levels in Mississippi Delta ecosystems, which suggests that the mid-South at least shared the cause of the problem leading to the decline. Additionally, with the advent of aquaculture in the region, the development and re-colonization of southern breeding populations may be enhanced. Wintering birds have certainly been able to regularly take advantage of catfish ponds for foraging and loafing. This steady resource has likely contributed to greater overwinter survival, enabling more birds to return to northern breeding grounds, and to remain in the south as year-round residents.

Figure 48. December estimates of DCCOs in the Southeastern U.S., Zone 4,* based on CBCs, 1984 – 1997.



* Includes AL, AR, FL, GA, KY, LA, MS, NC, OK, SC, TN, and TX; annual totals based on sites with ≥ 100 individuals

Table 5. Zone 4, Southern U.S. Summary of Population Data and Trends

(+ = increasing; - = decreasing; 0 = no change).

State	Status (B, M, W, w) ¹	Known colony sites/ time period	No. active col./last yr surveyed	No pairs/ last year surveyed	Trend (winter birds) ²	Winter number	Migrant number
AL	M, W	-	-	-	+	30,000-40,000 ³ ; X = 19,000 (7,000-33,000) ⁴	50,000
AR	B,M, W	1/1999	1/1999	72-100	+	X = 6200 (1,000-16,000) ⁴	1000s
FL	B, M, W	≥ 110/1989	84/1999	6,745 ⁵	+/?	X = 44,000 (28,000-67,000) ⁴	NA
GA	B, M, W	7/1990s	NA	NA	+	15,000-30,000 ³ ; X = 3176 (1500-6000) ⁴	NA
KY	M, w	-	-	-	+	500-1000 ⁴	100s-1000s
LA	B, M(?), W	2/1994-99	2/1994-99	427	+	X=19,000 (11,000-33,000) ⁴	NA
MS	B, M, W	2/1998	2/1998	33	+	65,000 ³ ; X=12,000 (1,000-23,000) ⁴	NA
NC	B, M, W	3/1980s-90s	NA	NA	+/?	25,000-50,000 ³ ; X = 50,000 (10,000-105,000) ⁴	NA
OK	B, M, W	2/1985-94	1/1996	30	+	X = 6600 (2,000-14,000) ⁴	NA
PR	w	-	-	-	0	Very few	-
SC	B, M(?), W	17/1985-96	10/1996	895	+	1000s ³ ; X=7,000 (2,000-17,000) ⁴	NA
TN	B, M, w	3/1992-99	1/1999	29	+/?	100s ³ ; X=235 (21-791) ⁴	2000
TX	B, M, W	≥ 5/1990s	NA	NA	+	X = 66,000 (50,000-90,000) ⁴	NA
Total		152	> 101	> 8231		NA	NA

1 = B, Breeding; M, Migrant; W, Wintering; w, small numbers winter

2 = Winter numbers reported for most states based on CBC data, which have not yet been analyzed, but precursory examination suggests increases over the last 10-15 years; for some states other data also support increasing trend (see summaries).

3 = Estimate from winter surveys conducted or reported by state, federal or non-governmental agencies

4 = Mean and Range based on number of individuals counted on Christmas Bird Counts conducted over the last decade.

5 = Numbers of individuals were estimated, but very wide range of possible numbers reported, due to inaccuracy of aerial survey method. See Florida summary for detail.

Figure 49. Distribution of South and Southeastern U.S. DCCO breeding population (Zone 4) 1980 – 1999.

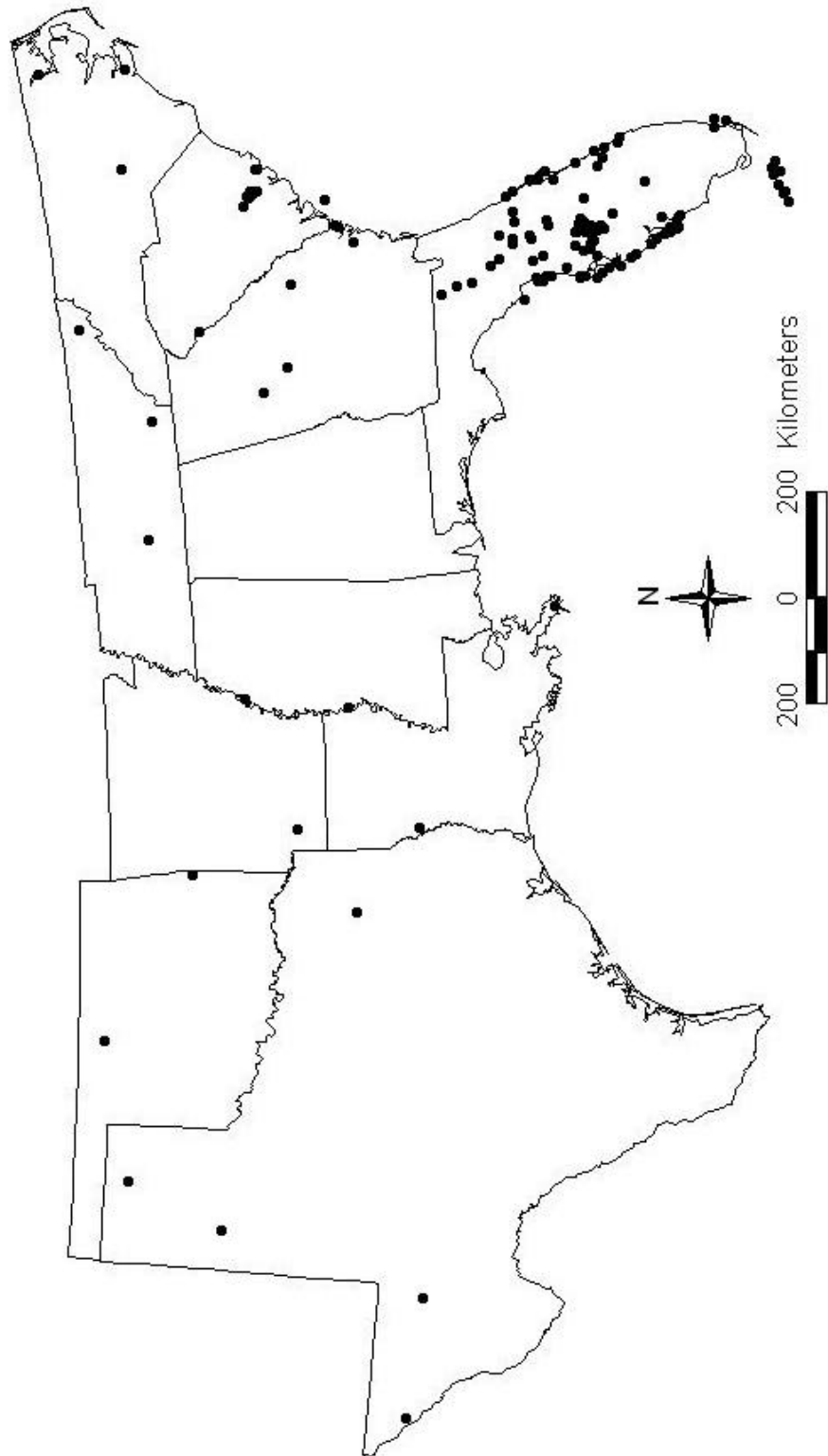


Figure 50. Distribution and size of active colonies in South Carolina during last complete survey (1996).

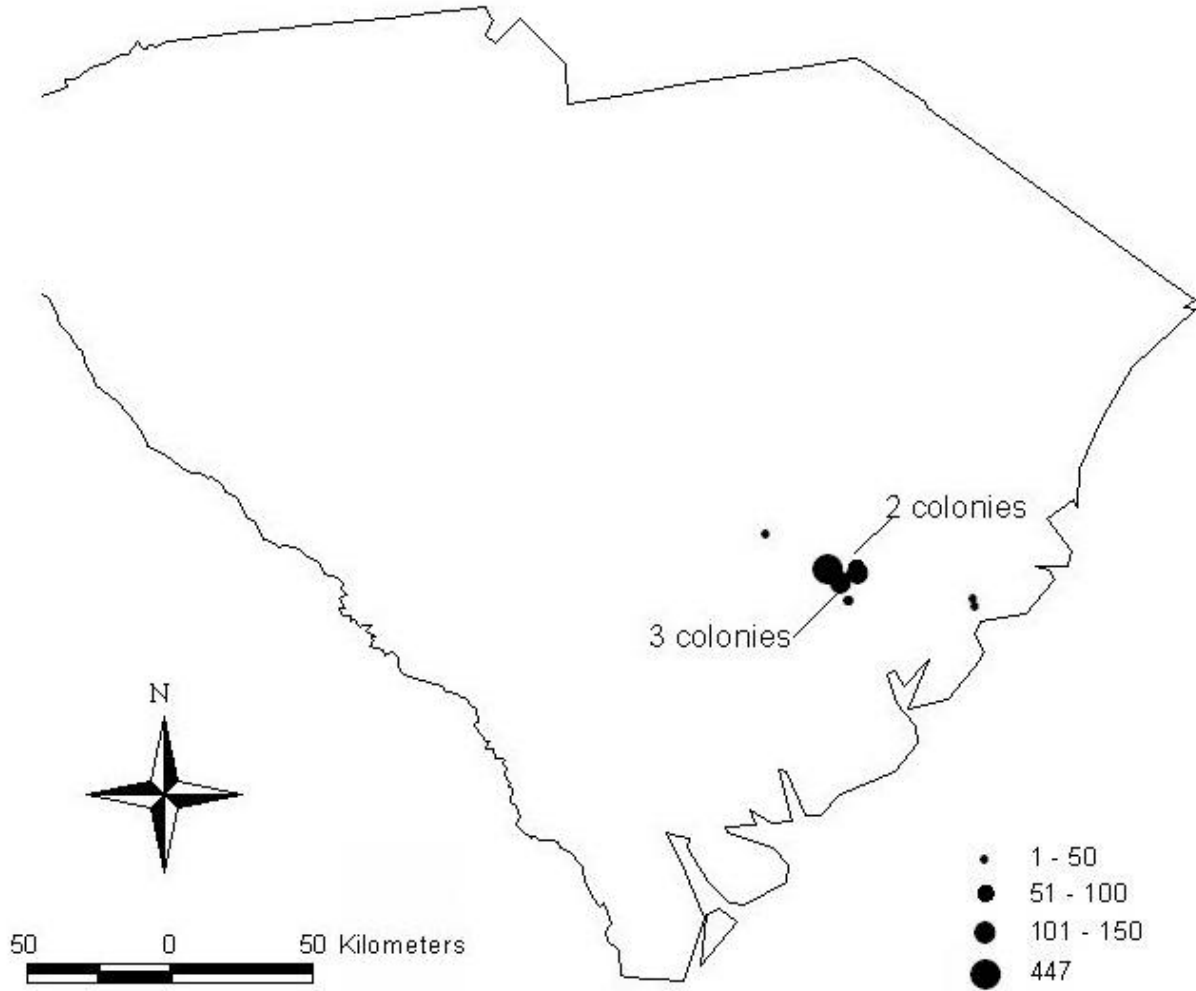
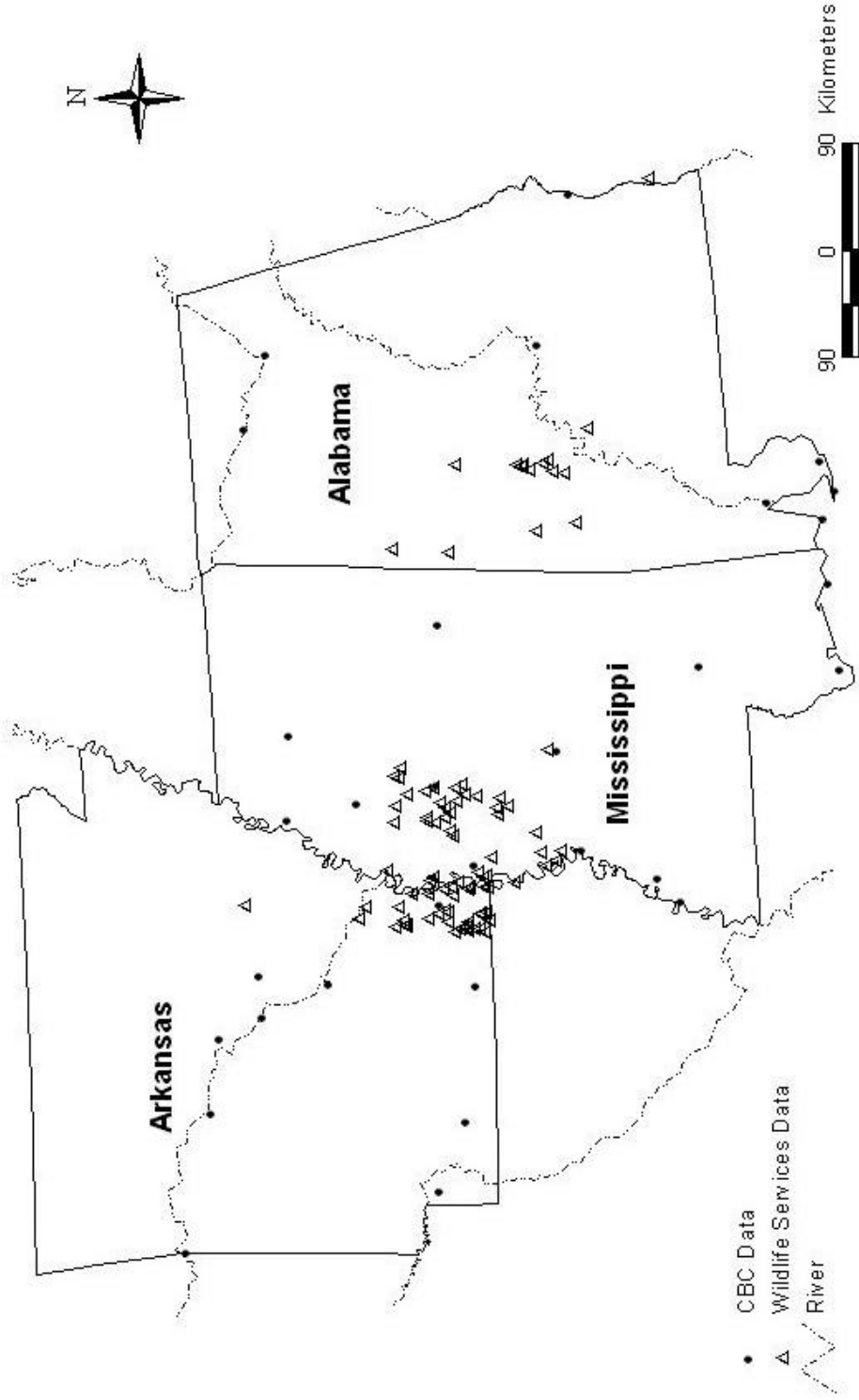


Figure 51. Comparison of Christmas Bird Count sites (with ³ 100 birds) with roost sites located by USDA/WS in Arkansas, Mississippi and Alabama.



Alabama

Summary of Population Data and Trends

(Survey completed by F. Boyd, USDA Wildlife Services, Auburn AL.)

Breeding.

The Double-crested Cormorant occurs throughout the year in Alabama, and is common October to May (Imhof 1976). However, it has not been documented as a breeder in the state (F. Boyd, pers. comm.).

Winter.

In winter, the number of birds fluctuates, and it is difficult to provide estimates of wintering vs. migrating birds, but large numbers occur. Roughly 30,000 – 40,000 individuals are thought to overwinter (F. Boyd, pers. comm.). There are 16 known sites where birds concentrate during both migration and winter (Figure 51); 15 of these sites are used for both feeding and roosting. Only one site in Gainesville was reported as strictly a feeding site (F. Boyd, pers. comm.). Numbers of wintering and migrant birds are abundant around the coast, and have been gradually increasing inland since the 1980s (see Historical Information, below); however, numbers may now be leveling off (F. Boyd, pers. comm.). Christmas Bird Counts conducted between 1959 – 1988 suggested an increase of 11.6 % per year. More recent Christmas Bird Counts (1989 – 1998) estimated that approximately 7,000 – 33,000 birds wintered in the state during the last decade. Though these data have not yet been analyzed, numbers appear to still be increasing, with high counts between 23,000 – 33,000 reported during 1995-1998 (Figure 52) (Sauer et al. 1996).

Migration.

While it is difficult to differentiate migrant from overwintering birds, in late winter large numbers of migrants begin to pass through. In March – April, peak numbers of 50,000 birds (combined migrants and wintering birds) are reported (F. Boyd, pers. comm.).

Historical Information

Little information available. Imhof (1976) reported that the species was abundant on the Gulf Coast in winter and during migration; it also occasionally summered on the coast. However, the species had begun to decline inland around 1945, and in 1961 was considered rare everywhere inland in the state (Imhof 1976). Numbers remained scarce inland until about 1980, at which time the species started showing up again in western Alabama. In the late 1980s, cormorants were observed in other areas of the state and along river systems. These were mostly migrants in northeastern Alabama and on the Eufala Reservoir in the southeast, where high concentrations are currently reported (F. Boyd, pers. comm.).

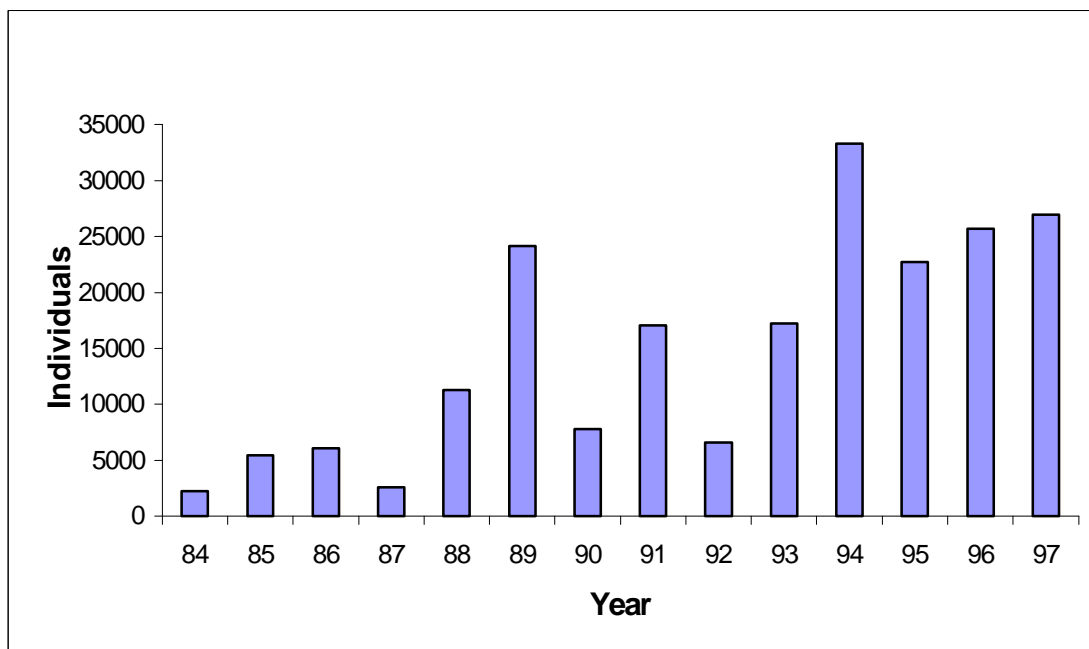
Land Ownership

Land ownership was provided for 11 of the 16 known roost / feeding sites. Six of these sites are on federal land, owned by the Army Corps of Engineers. The other five are on private lands, one of which is owned by a hunting club (F. Boyd, pers. comm.). No other information was provided.

Productivity

No documented breeding (F. Boyd, pers. comm.).

Figure 52. December estimates of DCCOs in Alabama based on CBCs, 1984 – 1997.



Arkansas

Summary of Population Data and Trends

(Surveys completed by M. Hoy, USDA/WS, Stuttgart, AR, and K. Rowe, Arkansas Game and Fish Commission, Humphrey, AR; winter data provided by S. Werner, USDA/WS, Mississippi State University, MS.)

Breeding.

In 1999, the Double-crested Cormorant was documented as a nesting species in Arkansas (Anonymous, 1999) for the first time since 1951 (James and Neal 1986), though unreliable reports of breeding have been made since that time (K. Rowe, pers. comm.). On May 19, 1999, more than 100 nests were discovered on Millwood Lake in southwestern Arkansas between Yarborough Landing and the dam. The birds were using only live Cypress trees. During this discovery, 11 cormorants were collected for DNA testing, food habits and other studies. On June 30, 72 active nests were documented and 137 adult and young cormorants were counted on the lake; 106 of these were collected for the previously mentioned studies, for a total of 117 birds collected (Hutchinson 1999). The state director for Wildlife Services stated that his agency

hoped to “keep this population down and maybe eliminate it,” as the catfish industry and sports fishermen do not want cormorants to become established year-round in Arkansas (Anonymous 1999).

Winter.

James and Neal (1986) reported that, in winter, the species is present locally in small numbers in the southern areas of the state, but that most birds leave after the arrival of severe temperatures. However, sometimes large numbers are still present in the latter half of December, especially in recent years. While no estimates of wintering numbers were available from state agencies, USDA/WS conducts winter cormorant surveys which are not state wide, but focus on catfish production areas. In 1991, 36 cormorant night roosts were located in southeastern Arkansas during February and March (Figure 51); numbers of cormorants observed at these roosts ranged from 1 to > 1000 individuals (S. Werner, pers. comm.). Additionally, during Christmas Bird Counts, large numbers have been tallied at reservoirs around the state. Counts conducted between 1985 – 1998 ranged between approximately 100 – 16,000 birds (Figure 53) (Sauer et al. 1996). Large increases appear to have occurred over the last decade, but percent change is not available, as these data have not yet been analyzed. In 1996, a high count of 16,000 birds was made; high numbers at Holla Bend NWR, Pine Bluff and Magnolia-Lake Columbia contributed most to this count. Jackson and Jackson (1995) examined Christmas Bird Count data from 1946 – 1991 and noted that counts in western Arkansas that included major reservoirs had increases paralleling those along the Mississippi River. During the last two Christmas Bird Counts (1997, 1998) numbers appear to have dropped considerably (Figure 53).

Migration.

Large numbers are recorded during spring and fall migration along the Arkansas River valley, Mississippi River bottoms, Bayou Meto bottoms, Millwood Lake, and at fish farming regions in the ne, se, and central portions of the state (M. Hoy, pers. comm.). In the fall, peak numbers are observed from mid-September to early-December. In the spring, peak numbers are recorded in February and March.

Historical Information

Audubon (1843) noted that in the early 1800s, shortly after young had fledged, many cormorants could be found as far north as the Arkansas River. In the late 1800s, the species was recorded as fairly abundant in parts of the state, and was apparently a year round resident. In the early part of 1889, cormorants were considered “fairly common in Poinsett County” (James and Neal 1986), suggesting that the species wintered in the state in fair numbers, at least during some years. Howell (1911) reported large numbers of migrants observed in the late 1800s and early 1900s at Helena, Menasha Lake and along the White River at Crocketts Bluff, but by the early 20th century, cormorants had apparently declined. By the time Howell was writing, the species could only be found in the wilder and more remote parts of the state, but was noted as being “formerly abundant in the rivers and swamps of eastern Arkansas.” Howell (1911) attributed the cormorant’s decline to human modification of the landscape. Though the first Arkansas nesting was not documented until 1910, the species apparently had nested in the state previously and in greater numbers; Howell (1911) reported that the colony of 100 – 200 nests in cypress trees that he observed in 1910 at Walker Lake, Mississippi County, was probably the only large colony that remained in the state at that time. By 1924, Walker Lake had been drained and the one large

nesting colony had disappeared. Prior to 1999, the last known nesting in the state was in 1951 at Grassy Lake (James and Neal 1986).

The species remained in the state mostly as a migrant, and large numbers were reported during spring migration as early as 1939, when 2000 were seen at Horseshoe Lake in February; 4000 were observed at Big Lake Refuge in March, 1946. Wintering numbers have recently increased; during Christmas Bird Counts between 1948 and 1961 few were reported. James and Neal (1986) suggested that the large numbers remaining in recent years may be attributable to the impoundment of large amounts of water and to recoveries from population declines noticeable in Arkansas by the 1960s.

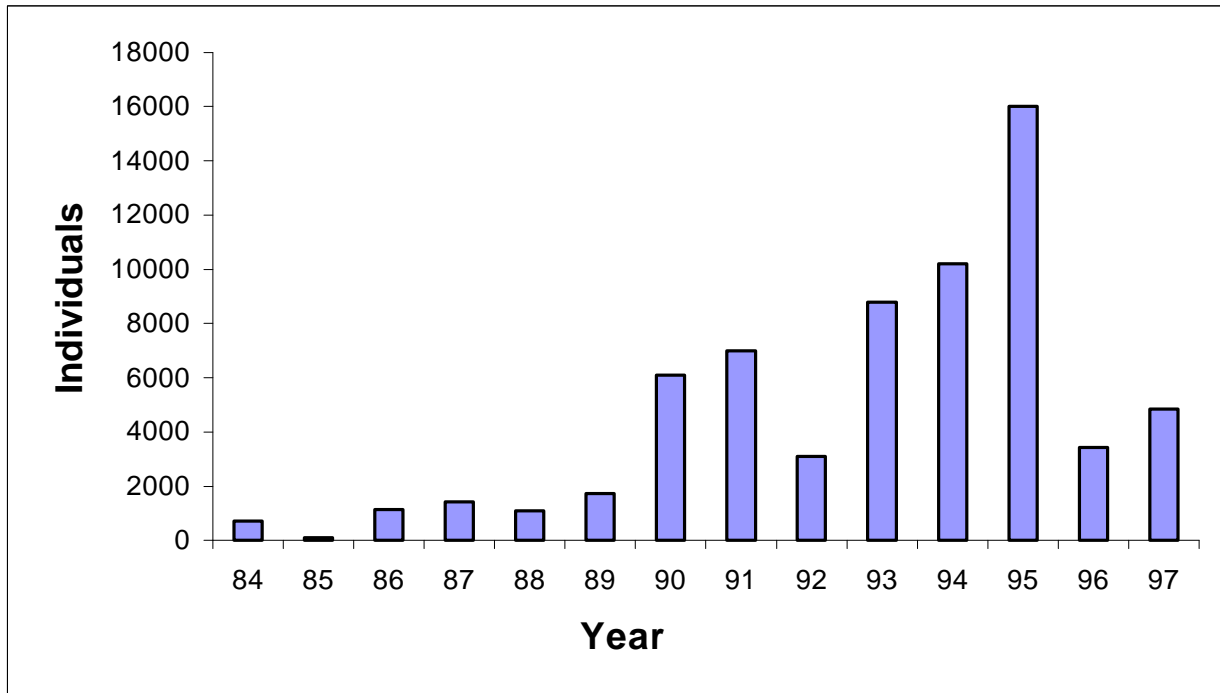
Land Ownership

Land ownership information was provided by K. Rowe and M. Hoy. K. Rowe provided information on nine sites: one wintering / migration site, Lake Millwood, and eight additional sites used during migration. Of these nine sites, three were on public lands owned by the Army Corps of Engineers; two were on state lands, and four were on private lands. M. Hoy provided information on wintering and migration sites at a broader, more general scale (e.g., Mississippi River bottoms, etc.). He noted that for the Arkansas and Mississippi River bottoms and Bayou Meto bottoms, where birds concentrate in winter and during migration, ownership was both public and private. Other wintering and migration sites were on private lands.

Productivity

No information provided.

Figure 53. December estimates of DCCOs in Arkansas based on CBCs, 1984 – 1997.





Florida

Summary of Population Data and Trends

(Survey completed by J. Rodgers, Jr., Florida Game and Fresh Water Fish Commission, Gainesville, FL.)

Breeding.

The Double-crested Cormorant is a year round resident in Florida. *P.a. floridanus* is the breeding race and is present year round; *P.a. auritus* migrates into the state for the winter. In 1999, the Florida Fish and Wildlife Conservation Commission conducted a statewide colonial waterbird survey and estimated there were 6,745 pairs (approximately 13,490 individuals) of Double-crested Cormorants at 84 colonies. Number of pairs at each specific colony was not available, thus we could not map colonies by size. However, the following data were provided: average colony size was 80.3 nests; 19 % of colonies were < 50 nests; 6 % of colonies ranged from 51 – 100 nests; 7.1 % ranged from 101 – 150 nests; only about 7 % of colonies had greater than 150 nests. Though estimates from statewide colonial waterbird surveys conducted in 1989 (8,898 – 23,980 individuals at 110 colonies) and 1977 (28,000 + individuals at 77 colonies) were also provided, at this time it is not possible to determine if cormorants in Florida are stable, increasing or decreasing (J. Rodgers, pers. comm.) for a number of reasons. The survey method consisted of flying over a colony and estimating the size class of the colony. At the end of the survey, size classes were tallied and numbers of birds summed up. The large range in the 1989 estimate is indicative of the difficulty of estimating the number of birds through aerial counts; dark plumaged species such as cormorants, it was noted, generate especially crude estimates that bias current and future population analyses. Additionally, the actual number of birds in 1989 was probably underestimated, because coverage of coastal sites was incomplete during this survey year. Finally, survey methods were modified in 1999 for better species detection and the data have not yet been analyzed. However, in 1989 only 84 colonies were located statewide, while in 1989, 110 colonies were counted (J. Rodgers, pers. comm).

Winter.

Large numbers winter throughout the state, but no information was provided on winter numbers. Christmas Bird Counts conducted from 1959 – 1988 suggested an increase in wintering birds of 3.5 % per year. More recent Christmas Bird Counts (1989 – 1998) estimated that approximately 28,000 – 67,000 birds wintered in the state during the last decade (Sauer et al. 1996). These data have not yet been analyzed, and trends are not readily apparent; however, high counts of approximately 67,000 and 54,000 birds were observed in 1992 and 1993, respectively (Figure 54).

Migration.

Cormorants migrate through the state; November – March was reported as the peak migration period, but no information was available on migration numbers (J. Rodgers, pers. comm.).

Historical Information

The Double-crested Cormorant has been a long time, year round and abundant resident in Florida. Audubon (1843) reported that the Florida Cormorant (*P. a. floridanus*) was a constant resident in Florida, the Florida Keys, and along the coast to Texas. The Double-crested Cormorant (*P.a. auritus*), he noted, rarely went farther south than the Capes of North Carolina. However, Howell (1932) noted that the Double-crested (*P.a. auritus*) occurred in the winter “not uncommonly,” south to Okeechobee Lake, Florida. He noted that “cormorants are moderately common in winter in Choctawhatchee Bay and all along the Gulf Coast; probably some of them are of this northern race.”

The earliest nesting record for Florida was obtained in April of 1832, when Audubon visited several of Florida’s small keys. Audubon (1943) reported large colonies of cormorants, with “many thousands of these birds,” and collected large numbers of birds and eggs. Nesting was also documented in the late 1800s, when egg sets were collected at Gainesville and at the Wakulla River (Stevenson and Anderson 1994). Around the first third of the twentieth century, Howell (1932) mapped the breeding distribution of the Florida Cormorant from the s. tip of the Peninsula north to the Chassahowitzka River and Merritt Island. He also showed a colony on the Wacissa River. Though he did not show colonies on several of the Florida Keys, he reported that breeding occurred there. Howell (1932) stated that the Florida Cormorant was “very abundant on the Gulf Coast, and less numerous on the east coast and on many lakes in the interior.” He reported that cormorants “may be seen in small or large flocks in all the bays and lagoons along the coasts, in most of the larger rivers and lakes, and in the ocean and Gulf. The abundance of the bird has led to the erection of a series of racks for many miles along the Gulf Coast to catch the excrement of the birds, which is collected to use for fertilizer.”

Howell (1932) also reported numerous observations of very large winter concentrations. On December 4, 1910, about 10,000 cormorants were seen at Passage Key. On January 21, 1919, a flock of 5,000 – 6,000 was observed at Terra Ceia Bay. Great flocks of cormorants were also observed in March 1925, along with ibises and herons, flying from the mainland at Cape Sable to outlying keys to night roosts. Additionally, Pangburn (1919) estimated 12,000 cormorants were at Pass-a-Grille on February 11, 1918.

Sprunt (1954) revised Howell (1932) and noted that for both subspecies, “no change of any consequence has taken place ... since Howell wrote in 1932.”

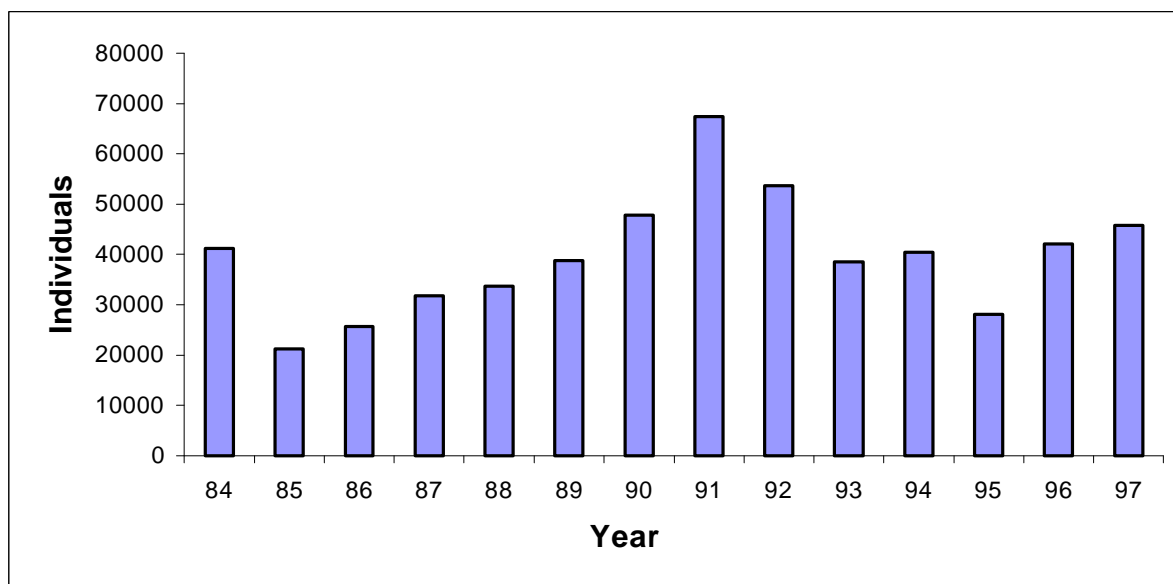
As discussed above, increases in breeding numbers may have occurred, at least in the last few decades. Additionally, very large winter concentrations have been recorded in the 1980s and 1990s; Christmas Bird Count maxima include 15,100 at North Pinellas county, December 22 1984, and about 25,000 off Anclote Key, January 15, 1991 (Stevenson and Anderson 1994).

Land Ownership

No information provided.

Productivity

No information provided.

Figure 54. December estimates of DCCOs in Florida based on CBCs, 1984 – 1997

Georgia

Summary of Population Data and Trends

(Survey completed by B. Winn, Georgia Dept of Nat Resources, Coastal Nongame Office, Brunswick GA.)

Breeding.

The Double-crested Cormorant is a recent breeder in Georgia; first nesting was documented in May, 1990 at Blalock Lake (Brisse 1990). Since that time a total of seven colonies has been discovered, but no statewide surveys for cormorants have been conducted, so the total number nesting in the state is not known (B. Winn, pers. comm.). Recently, cormorants have been found in increasing numbers inland, especially in the Piedmont and along the “fall line,” mostly in the Macon and Augusta areas (Brisse 1990).

Wintering.

Large numbers of birds, estimated between 15,000 – 30,000 winter in the state, roosting and feeding at sites along the Lower Altamaha River and Delta, at lakes and reservoirs, and along the coast (B. Winn, pers. comm.). Christmas Bird Counts conducted between 1959 – 1988 suggested an increase of 5.9 % per year. More recent Christmas Bird Counts (1989 – 1998) estimated that approximately 1,000 – 6,000 birds wintered in the state during the last decade (Figure 55) (Sauer et al. 1996). This is likely an underestimate; however, the CBC data appear to

support an increasing trend, especially since the mid 1980s, though these data have not yet been analyzed and percent change is not available.

Migration.

During migration, consistent large flocks are observed along the entire coast, from Tybee Island to Cumberland Island, but no estimate of numbers is available. In spring, peak numbers occur in March and April; in fall, peak numbers are reported in November – December (B. Winn, pers. comm.).

Historical Information

The only Historical Information we were able to obtain for Georgia was from records published in Burleigh (1958). The earliest record in the state was of a specimen taken at Summerville, Chattooga County, on March 1, 1886; this individual was identified as *P. a. floridanus*. In the early part of the twentieth century, the species was fairly abundant in the winter at various locations within the state. Non-breeding summering birds were documented in 1939. At the time of Burleigh's (1958) writing, the cormorant was characterized as "resident on the coast, occurring as a common winter visitant, and in smaller numbers throughout the summer months." In the interior, it was a "fairly common" winter resident in the southern portion of the state. Since that time, the population, particularly in the winter, has obviously increased.

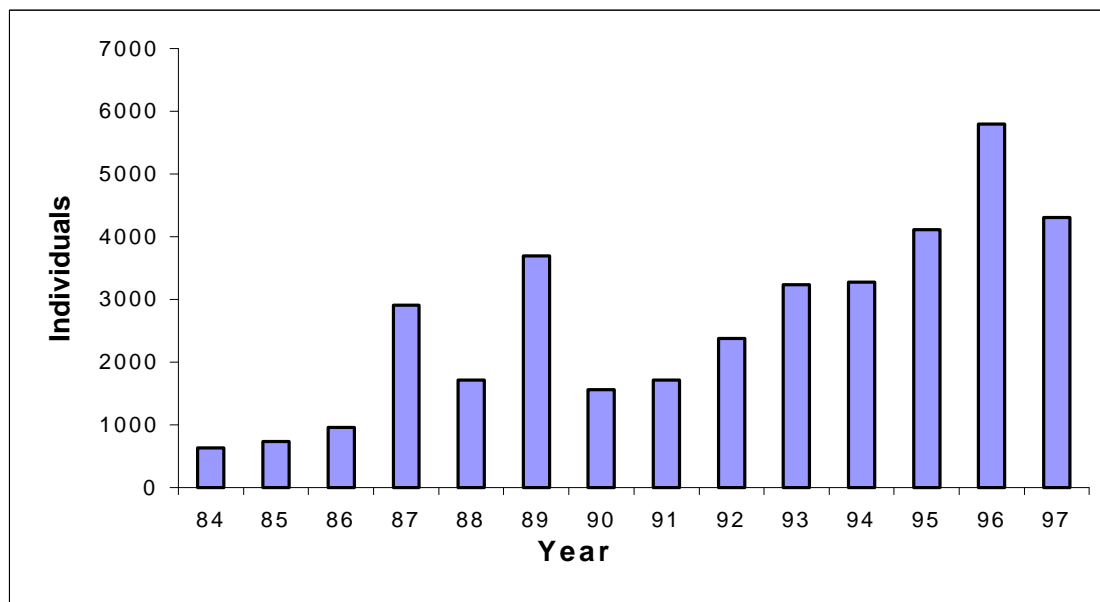
Land Ownership

Land ownership was provided for wintering sites along the Lower Atlanta River and Delta, where cormorants roost and feed on state, federal and private land; and for migration areas along the coast, which is mostly state and federal land (B. Winn, pers. comm.).

Productivity

No information provided.

Figure 55. December estimates of DCCOs in Georgia based on CBCs, 1984 – 1997.





Kentucky

Summary of Population Data and Trends

(Survey completed by B. Palmer-Ball, Jr., Kentucky State Nature Preserve's Commission, Frankfort, KY.)

Breeding.

The Double-crested Cormorant has not been documented to breed in Kentucky since 1951 (Palmer-Ball 1996).

Winter.

Depending on the severity of winter weather, up to 1000 birds will winter in the state. Numbers are believed to have increased recently, since the late 1980s. In 1998, 919 birds were counted during the CBC conducted at Lake Barkley (Sauer et al. 1996), the only regular wintering area in the state, where birds both roost and feed.

Migration.

During migration, birds are observed regularly at Lake Barkley and at Reelfoot Lake, though birds at this latter site are probably flying from roost sites at Reelfoot Lake in Tennessee. Migrants also sometimes occur on the Ohio River at Markland Dam. In the spring, peak numbers are observed in April – May; in the fall, peak numbers occur October-November. The highest count recorded was 2400 at Markland Dam, Nov 6, 1997, but birds are usually present only in the hundreds at any one location (B. Palmer-Ball, pers. comm.).

Historical Information

Palmer-Ball (1996) notes that the Double-crested Cormorant was not documented to have nested in Kentucky prior to the twentieth century. The earliest records we were able to obtain for the species in the state were those reported by Pindar (1925), who made observations of the birds of Fulton County, Kentucky, between 1890 – 1893. Based on his observations, Pindar (1925) noted that the Double-crested Cormorant (*P. a. auritus*) was “a common migrant, a fairly common summer habitant, and rare in winter” in Fulton county. He considered the Florida Cormorant (*P.a. floridanus*) “a rare summer visitant,” but noted that it probably bred at Reelfoot Lake, and possibly at other suitable locations.” Mengel (1965) reported three colonies in the late 1930s in the western portion of the state. In 1949, Mengel visited one of these colonies (Fulton County) and observed about 50 nests. Mengel (1965) described the state breeding distribution as “the Ohio River bottom lands from Henderson westward, through much of the low country of the Purchase, especially in Fulton, Hickman, Carlisle, and Ballard counties, the species occurs locally but in numbers near colonies [the three mentioned above] and in localities where unknown colonies probably exist.” Mengel (1965) also mentions that he saw cormorants in June and July on each of many visits he made to Swan and Clear Lakes in Ballard County.

Mengel (1965) reported that in winter cormorants were “rare and irregular” except near Reelfoot Lake, where they were common year round. He noted that a few birds probably wintered around Kentucky Lake and on the lower Ohio river.

Land Ownership

Land ownership was provided for Lake Barkley, which is federal land owned by the Army Corps of Engineers; and for the Reelfoot Lake area, where birds roost and feed on privately owned land (B. Palmer-Ball, pers. comm.).

Productivity

No information provided.



Louisiana

Summary of Population Data and Trends

(Survey completed by B. Vermillion, Louisiana Dept of Wildlife and Fisheries, Baton Rouge, LA.)

Breeding.

The number of Double-crested Cormorants breeding in Louisiana appears to be increasing, but it is not known by how much. Prior to the late 1980s – early 1990s, the species was only known to nest in small numbers in the state, but since that time larger colonies have been noted. Presently, only two breeding sites are known. One is at the mouth of the Mississippi River; in 1994 this colony had about 50 pairs, and in 1999 about 30 pairs nested there . The other colony is on the Toledo Bend, a large reservoir on the Louisiana / Texas border; numbers were counted in 1995 only, at which time 397 nests were found scattered in dead trees throughout the Bend (B. Vermillion, pers. comm.).

Winter.

Large numbers winter in the state. Christmas Bird Counts conducted between 1959 – 1988 suggested that numbers increased by 17.4 % per year. More recent counts (1989 – 1998) estimated that approximately 11,000 – 33,000 birds wintered in the state during the last decade (Figure 56) (Sauer et al. 1996). Though these data have not yet been analyzed, numbers appear to still be increasing, with high counts of approximately 27,000 and 33,000 birds observed in 1996 and 1998, respectively. In the latter year, an unusually high count of nearly 19,000 birds was made at Shreveport; the previous high count for Shreveport occurred in 1996 with 9,264 birds observed.

Migration.

It is not known if cormorants pass through or stop over in Louisiana during migration; it is assumed that birds observed in the state are wintering, but possibly migrants from Texas going back to northern breeding grounds may pass through Louisiana (B. Vermillion, pers. comm).

Related Species.

Louisiana is one of only two states (the other is Texas) where substantial numbers of Neotropic Cormorants nest and overwinter (Telfair and Morrison 1995). Because the Depredation Order is in effect in these states, special efforts should be made to distinguish the Neotropic from the Double-crested Cormorant; in the U.S. numbers of Neotropics, while increasing, are much lower than those of Double-crested Cormorants. Neotropic cormorants likely feed in aquacultural areas in southwestern Louisiana (C. Hunter, pers. comm.) and are shot because of perceived impacts to fish, but extent to which shooting has or may affect this species is unknown (Telfair and Morrison 1995).

Historical Information

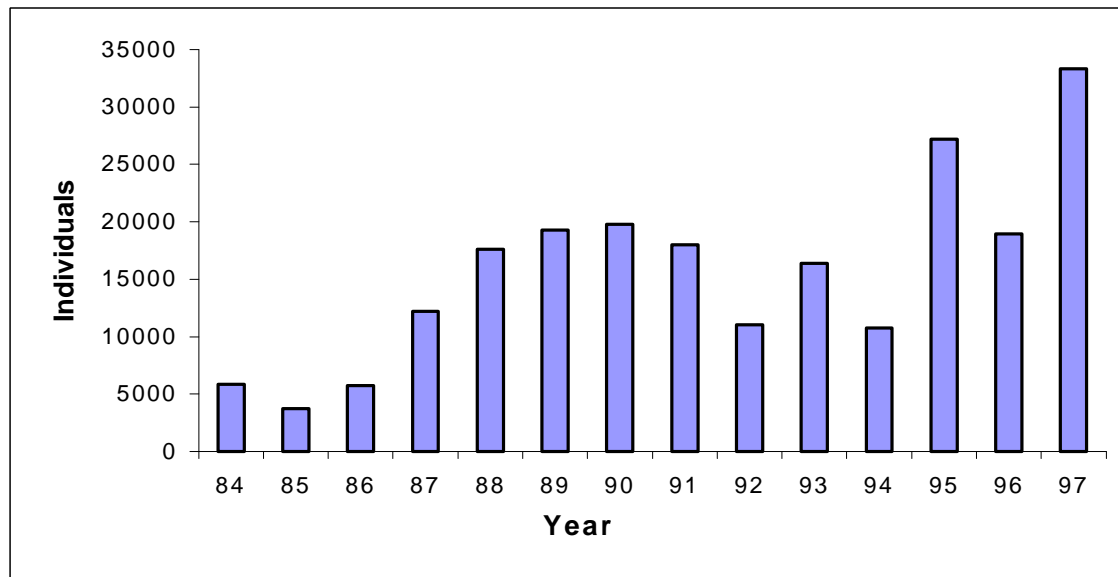
Audubon (1843) recorded *P. a. floridanus* as “constantly resident in the Floridas and their Keys, and along the coast to Texas”, thus we assume the species was present along the Louisiana coast and probably nested there at the time of Audubon’s visit. Audubon (1843) reports seeing cormorants “offered for sale in the New Orleans market, the poorer people there making gumbo soup of them.” However, it is not clear if these cormorants were obtained in Louisiana or if they were shipped into the state from Florida. Beyer et al. (1907) described the Double-crested Cormorant (*P.a. auritus*) as an uncommon winter resident in Louisiana, found mostly along the Gulf Coast as late as April. The Florida Cormorant (*P.a. floridanus*) occupied the same localities, but was much more common and bred everywhere within the state that it occurred; its distribution and abundance remained uniform throughout the year. Oberholser (1938) documented nesting in southwest Louisiana in Cameron Parish, in West Feliciana Parish and in Devil’s Swamp north of Baton Rouge, East Baton Rouge Parish. However, by the time Lowery (1974) revised his “Louisiana Birds”, there was no longer any evidence that the species still nested in the state. Though Lowery (1974) noted that cormorants were most common in Louisiana in the winter and early spring, he remarked that cormorants appeared less abundant in all seasons than they had been during the 1930s – 1950s. He also noted that, while the cormorant “used to be a winter fixture at City Park and University lakes in Baton Rouge, now its appearance on these lakes is unusual.” Apparently, numbers began increasing sometime after this period.

Land Ownership

No information provided.

Productivity

No information provided.

Figure 56. December estimates of DCCOs in Louisiana based on CBCs, 1984 – 1997.

Mississippi

Summary of Population Data and Trends

(Survey completed by J. Glahn, USDA Aphis/Wildlife Services, Mississippi State, MS.)

Breeding.

There are only three breeding records for the Double-crested Cormorant in Mississippi. The first was reported from a cypress brake near Clayton in May, 1952; 12 nests were observed (Coffey 1952). The latter two records were both obtained in 1998, with two nests observed at Lewis Swamp, and 31 nests observed at Jones Lake (Reinhold et al. 1998).

Winter.

Very large numbers winter in the Delta Region of Mississippi. USDA/WS conducts yearly winter cormorant surveys which are not state wide, but focus on catfish production areas. In 1999, there was a total of 76 known night roosting sites (Figure 51), and 64,169 birds were counted. A high count of 68,179 birds was recorded in 1998. Results from annual censuses conducted in February, 1990 – 1999, show dramatic increases, from 28,584 birds in 1990 to the present numbers. A low of 16,405 birds was counted in 1994 due to icing over of the majority of fish ponds in the Delta (J. Glahn, pers. comm.). Christmas Bird Counts conducted between 1959 – 1988 suggested an increase of 18.7 % per year, the largest increase reported for any of the wintering states. More recent Christmas Bird Counts (1989 – 1998) estimated that approximately 1,000 – 23,000 birds wintered in the state during the last decade (Figure 57) (Sauer et al. 1996).

While these counts are obvious underestimates (see Summary for Zone 4: Summary of Population Data for Southeastern U.S.), and these data have not yet been analyzed for percent change, they appear to support an increasing trend since the mid-1980s.

Migration.

Peak migration numbers occur in March, but no estimates are available (J. Glahn, pers. comm.).

Historical Information

Jackson and Jackson (1995) described the history of the Double-crested Cormorant in Mississippi, which we have summarized here, unless otherwise noted. In the mid 1800s, cormorants were abundant in late summer and early fall along the Yazoo and Mississippi rivers. Lewis (1929) reported that interior nesting birds wintered along the coast from western Florida to eastern Texas, and in the first half of the 20th century, Burleigh (1944) reported it as one of the most common winter birds of the Mississippi Sound, late September to early May. In freshwater areas, such as the lower Mississippi River, Lewis (1929) reported that winter numbers were very small.

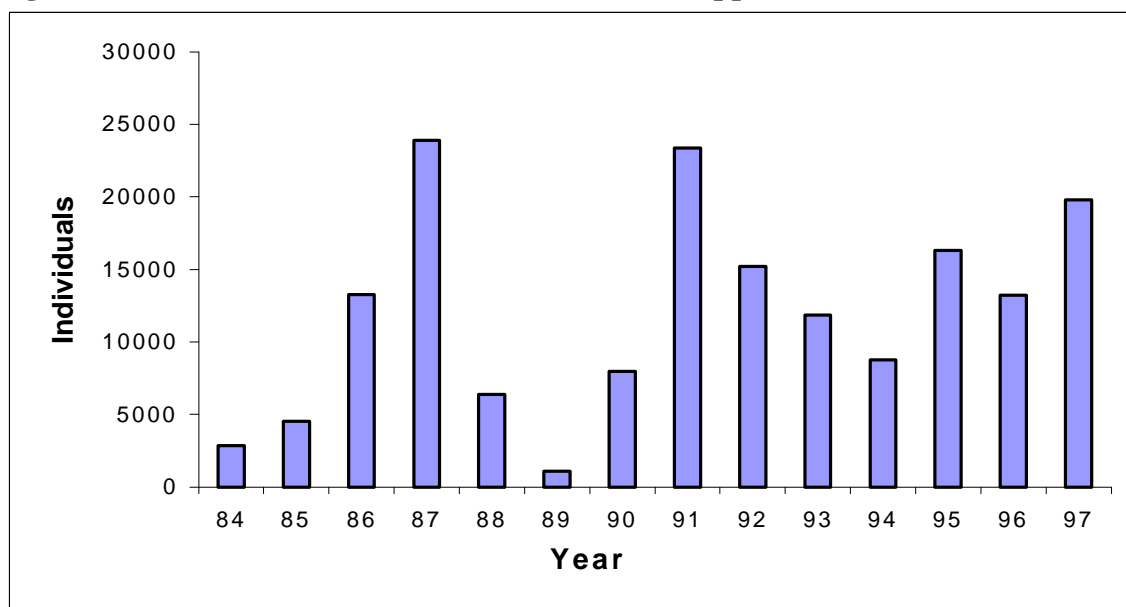
At the time of the first breeding in the early 1950s, tenant farmers in the Clayton area reported that there had been many more birds, but that much cypress had been cut. A pattern of decline from the 1940s – 1970s occurred throughout the south central U.S., and is thought to be linked to DDT usage and persecution on both the breeding and wintering range. It was during this declining point that the catfish industry underwent its major growth in the mid-South, with the heart of the catfish industry located in the Mississippi Delta at the confluence of the Arkansas and Mississippi rivers, both major migratory pathways for many avian species. Cormorants began recovering in the south in the late 1970s, and since that time, the number of wintering cormorants has increased substantially, with larger numbers appearing to arrive earlier and stay later each year. The species is now regularly found on the Mississippi coast year round, although substantial numbers are reported only in winter.

Land Ownership

The two breeding sites documented in 1998 were both on private lands; Jones Lake is owned by a private individual, and Lewis Swamp is owned by a private corporation. Nineteen of the 76 night roosting sites used by wintering birds are on public agency lands; the other 57 are on lands owned by private individuals (J. Glahn, pers. comm.).

Productivity

Of the 31 nests on Jones Lake, 13 fledged one or more young (Reinhold et al. 1998).

Figure 57. December estimates of DCCOs in Mississippi based on CBCs, 1984 – 1997.

North Carolina

Summary of Population Data and Trends

(Survey completed by D.H. Allen, North Carolina Wildlife Resources Commission, Trenton, NC.)

Breeding.

The Double-crested Cormorant has been nesting in North Carolina for at least two and a half centuries, though not consistently (see Historical Information, below). In the mid 1980s and 1990s, breeding was documented at three sites within the state: Great Lake / Lake Ellis Simon near the coast; along the Chowan River; and in the Piedmont at Jordan Lake, near Raleigh. In 1996, only the Great Lake / Lake Ellis Simon site was active, and 135 pairs were counted (D. Allen, pers. comm.); at this time, cormorants had shifted from Great Lake to nearby Lake Ellis Simon [which is 0.8 km southeast of Great Lake (Doig et al. 1989)]. Cormorants are no longer breeding at the Chowan River site. No data are available on the number of pairs from 1996 – 1998 at the Piedmont site, but in 1999, 50 breeding pairs were estimated (W. Golder, pers. comm.). No trend information is available, as most colonial waterbird survey efforts in North Carolina have focused on coastal nesters (D. Allen, pers. comm.).

Winter.

The wintering population is thought to be very large, estimated at 25,000 – 50,000 birds, possibly more, depending on the severity of the winter. Most birds winter on the inlets of Pamlico Sound, with the largest numbers (10,000 – 30,000) at Hatteras Inlet and Oregon Inlet (H. LeGrand and W. Golder, pers. comm.). Substantial numbers also winter at Beaufort Inlet and at Bald Head Island, Cape Fear; smaller numbers also winter at some inland sites (H. LeGrand, pers. comm.). Christmas Bird Count data from 1959 – 1988 suggest an increase of 9.9 % per year. More recent Christmas Bird Counts (1989 – 1998) estimated that approximately 10,000 – 105,000 birds wintered in the state during the last decade (Figure 59) (Sauer et al. 1996), but these data have not yet been analyzed, and recent trend information is not available.

Migration.

No information available on migration numbers or peak periods; since large numbers winter in the state, it is difficult to distinguish migrants (H. LeGrand, pers. comm.).

Historical Information

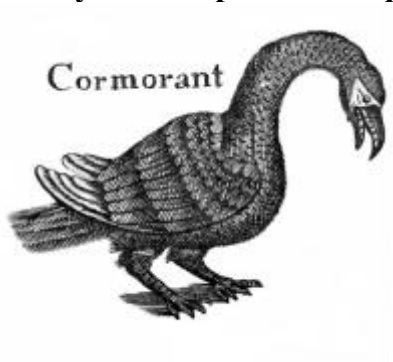
The Double-crested Cormorant appears to have been a very abundant breeding species in North Carolina, and possibly the surrounding region, during the colonial era. The earliest record we obtained for the state was from Brickell (1737), cited in Lewis (1929); Lewis noted that this record “doubtless included *P.a. floridanus* and *P.a. auritus*.” Brickell reports:

The Cormorants are the same as in *Europe*, only those of this Province are larger. They are as numerous all over these Parts of America, as in any part of the World, especially at the run of the Herrings, which is in *March* and *April*; at which time they are seen sitting upon the Sand Banks, or Logs of Wood in the Rivers, and catch vast quantities of Fish, which is their only Food, and whereof they are very ravenous and greedy. They lay their Eggs in the beginning of the Spring, in the Islands, in the Sound and near the Sea Shoar in the Banks, and sometimes on high trees, as the Shags do...”

Lewis (1929) also noted that Brickell (1737) included a representation of the Cormorant of North Carolina, which is the earliest representation of *P. auritus* he had seen (Figure 58). The next record we obtained for the state was from Audubon (1843), who reported only that, after young fledge, many proceed as far south as the Capes of North Carolina. He did not mention nesting, and numbers apparently declined; nesting was not documented in the state again until 1898, when 150 nests were counted on the coastal plain at Great Lake, Craven county (Pearson et al. 1959). This colony was visited again in 1909 and 1911, at which times 123 and 159 nests, respectively, were counted. Pearson et al. (1959) noted that the colony shifted its nesting site several times as the trees that it occupied were killed by the birds’ excrement. The colony appears to have slowly declined after this time, and in 1939, only 41 nests were counted. At the time of the first printing of Pearson et al. (1942), the authors noted that this was “the only known colony of breeding cormorants between Florida and Maine.” [However, breeding was documented in Massachusetts in 1940, and was reported to occur as early as 1937, and perhaps earlier (see Massachusetts summary)]. In 1948, 35 nests were counted at Great Lake. In 1953, nesting was recorded at Lake Ellis. In 1956 there was no nesting at Great Lake and only nine nests recorded at Lake Ellis (Pearson et al. 1959). In 1975, 12 nests were reported at Great Lake / Lake Ellis. Breeding numbers began to slowly increase in the 1980s, when a total of 74 pairs were counted at three sites (D. Allen, pers. comm.).

Pearson et al. (1959) note that most, if not all, cormorants seen in the state in winter were the *P.a. auritus* subspecies. These authors also provide records of birds taken in the fall between 1923 – 1938.

Figure 58. Double-crested Cormorant from Brickell (1737), possibly the first published representation of this species.



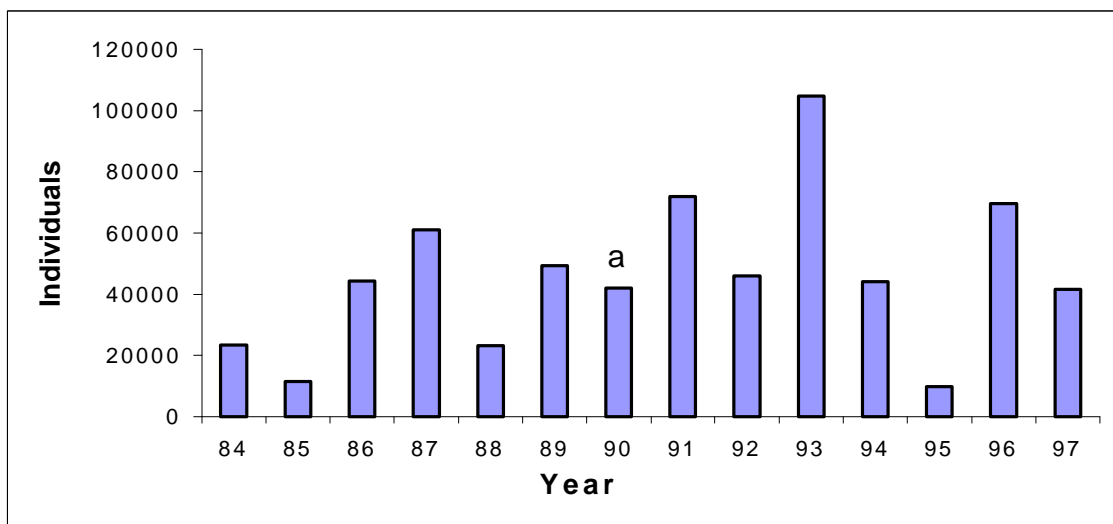
Land Ownership

The nesting site on the Chowan River is owned by a private timber company. The nesting site on Great Lake is federal land, part of the Croatan National Forest. The nearby Lake Ellis site is on privately owned land. The Jordan Lake site is owned by the federal government and administered by the Army Corps of Engineers. Wintering areas along the coast are primarily owned by the federal and state government, though there is also some private ownership (W. Golder, H. LeGrand, and D. Allen, pers. comm.).

Productivity

No information provided.

Figure 59. December estimates of DCCOs in North Carolina based on CBCs, 1984 – 1997.



a = In 1991, counts for Portsmouth and Ocracoke Islands gave very large numbers that appeared incorrect, thus data for these islands in this year were thrown out and replaced with the yearly average for the last 10 years (G. LeBaron, pers. comm.).



Oklahoma

Summary of Population Data and Trends

(Survey completed by M. Howery, Oklahoma Dept of Wildlife Conservation, Oklahoma City, OK.)

Breeding.

The Double-crested Cormorant currently nests at the Salt Plains Reservoir, Alfalfa County; it is the only known nesting site for the species in the state of Oklahoma. The discovery of breeding birds at this site in 1994 was the first breeding record for the state since 1985. The Salt Plains Reservoir colony was last estimated at about 30 pairs in 1996 (M. Howery, pers. comm.).

Winter.

The species overwinters on reservoirs in the state, but no winter surveys have been conducted. However, USDA/WS conducted a year-long wildlife assessment on Lake Overholser in Oklahoma City, for Will Rogers World Airport, from February 1997 to January 1998. During that time cormorant numbers fluctuated with seasonal migrations, but were most noticeable from February to April. Average counts for February, March, and April were 118, 428, and 355, respectively. Although the counts were averaged, during one count in April 1350 cormorants were observed foraging for fish. Additionally, during the same time period, 3,000 – 4,000 cormorants were observed roosting at Lake Arcadia, approximately 10 miles east of Lake Overholser. This could indicate a much larger number of cormorants wintering in Oklahoma than was previously known (P. Robinson, pers. comm.). Anglers believe cormorant numbers and the extent of their residency in the state have increased (Simmonds et al. 1997). Christmas Bird Counts conducted between 1959 – 1988 suggest an increase of 16.5 % per year during this period. More recent Christmas Bird Counts (1989 – 1998) estimate that approximately 2,000 – 14,400 birds wintered in the state during the last decade, with high counts occurring 1991 – 1993 (Figure 60) (Sauer et al. 1996). Though these data have not yet been analyzed, they suggest that numbers may still be increasing.

Migration.

During migration, large numbers occur, with peak fall numbers recorded October – early November; peak spring numbers are recorded March – April (M. Howery, pers. comm.). At most reservoirs where birds are observed, densities are greatest in fall migration, followed by spring migration and winter residency (Simmonds et al. 1997).

Historical Information

The earliest record we obtained of the species in the state was of a flock reported in fall 1916, apparently during migration (Nice 1931). The earliest nesting record we found was for a small colony (3 – 7 nests) at the north end of the reservoir in the Salt Plains National Wildlife Refuge between 1945 – 1950 (Sutton 1967; Baumgartner and Baumgartner 1992). Later, the species bred at the Robert S. Kerr Reservoir at the Sequoyah National Wildlife Refuge from 1973 – 1980 (M. Howery, pers. comm.) and again at the Sequoyah NWR in 1985 (Baumgartner and Baumgartner 1992). It was not recorded breeding again until 1994 when it returned to the Salt Plains Reservoir (M. Howery, pers. comm.).

Sutton (1967) recorded the species as transient throughout the state September to December, and from March to May, and noted that there were only three January records, and no February record. However, during the 1984 CBC, 2,100 birds were counted at the Sequoyah National Wildlife Refuge (Baumgartner and Baumgartner 1992). We are not sure when winter numbers began to increase, or whether these data actually represent an increase, since we found very little information on winter numbers and distribution in the state.

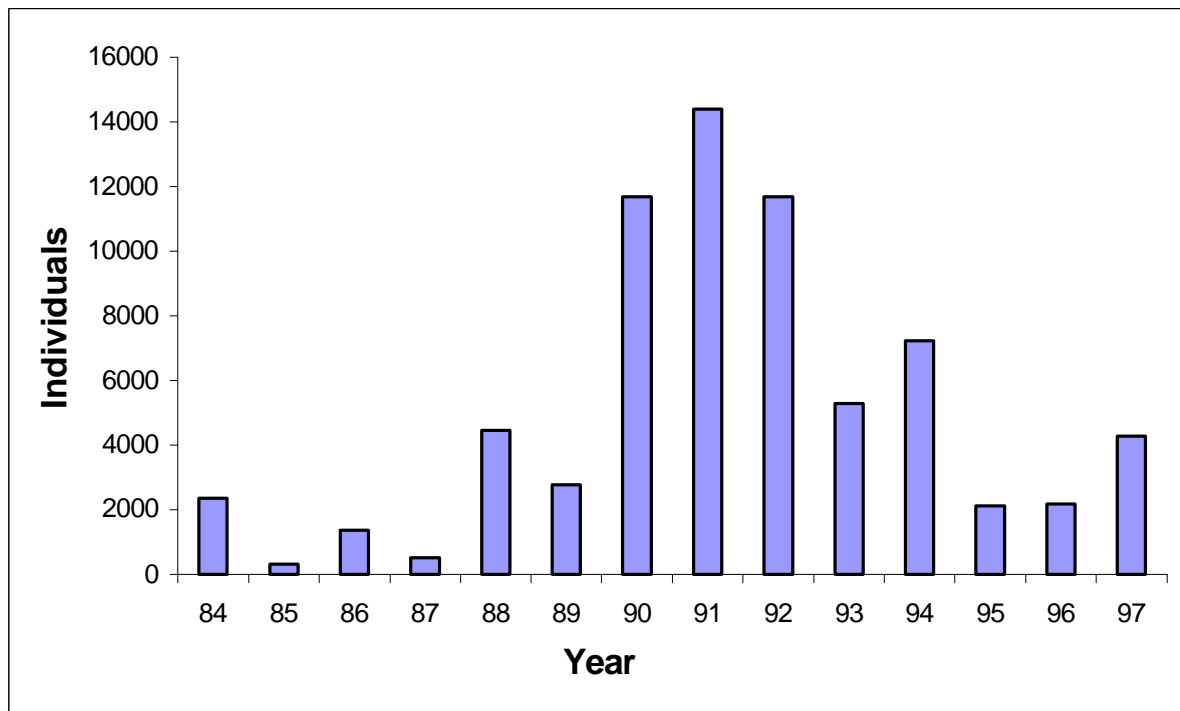
Land Ownership

Both breeding sites are on federal lands that are part of National Wildlife Refuges (M. Howery, pers. comm.).

Productivity

In 1974, 203 young were produced in 86 nests at the Robert S. Kerr Reservoir, giving a rate of 2.4 chicks per nest (M. Howery, pers. comm.).

Figure 60. December estimates of DCCOs in Oklahoma based on CBCs, 1984 – 1997.





Puerto Rico / Virgin Islands

Summary of Population Data and Trends

(No survey completed.)

In the eastern Caribbean, the Double-crested Cormorant occurs only as a winter vagrant. Cuba is the only Caribbean location that may have larger numbers of cormorants (J. Saliva, pers. comm.).



South Carolina

Summary of Population Data and Trends

(Survey completed by T.M. Murphy, South Carolina Dept of Natural Resources, Green Pond, SC.)

Breeding.

The Double-crested Cormorant was documented as a breeding species relatively recently in South Carolina, with the first nesting recorded in 1985. There are currently 17 known colony sites, all inland; during the last statewide count, 10 colonies were active. Complete ground counts for all known colonies were conducted in 1995 and 1996, and 717 and 895 pairs, respectively, were estimated. Complete ground counts were also conducted in 1989 – 1990, when 186 and 126 pairs, respectively, were counted. The number of breeding pairs is definitely increasing (Figure 61), but it is not known at what rate, because complete censuses have not been regularly conducted; additionally, some colonies were not known at the time complete censuses were conducted, so number of pairs may be underestimates. In 1998, 725 pairs were counted, but data were available for only five colonies (T. Murphy, pers. comm.).

Winter.

Large numbers winter in the state, feeding and roosting at inland lakes and reservoirs, and along the entire coast. In the last decade the population has increased along the coast. Christmas Bird Counts conducted between 1989 – 1998 estimate that approximately 2,000 – 17,000 birds wintered in the state during the last decade (Figure 62). While these data have not yet been analyzed for trends, numbers appear to be increasing (T. Murphy pers. comm.).

Migration.

It is not known if areas in the state provide important stopover sites for migrants, or how many migrants pass through, probably because large numbers winter in the state and it is difficult to distinguish migrant from wintering birds (T. Murphy pers. comm.).

Historical Information

The earliest record we obtained in the state was that of Hoxie (1892, cited in Lewis 1929) who observed cormorants foraging off the coast of South Carolina in winter. Sprunt and Chamberlain (1949) described *P.a. auritus* as an abundant winter resident, mostly coastwise. Wayne (1910) reported that the Florida Cormorant (*P.a. floridanus*) bred in the state, but he was never able to locate a colony. Sprunt and Chamberlain (1949) reported that the Florida Cormorant was a fairly common permanent resident, chiefly along the coast and that it was presumed to breed there. However, W. Post (pers. comm.) noted that cormorants probably did not breed in the state until the large hydroelectric lakes were built over the vast Santee Swamp. The first nest was finally found on L. Marion, about 55 mi from the coast, in 1985.

Land Ownership

No information provided.

Productivity

Post and Seals (1991) reported a rate of 2.20 fledglings per nest at Hog Swamp, a newly established colony, in 1989.

Figure 61. Changes in number of breeding pairs in South Carolina, 1989 – 1996.

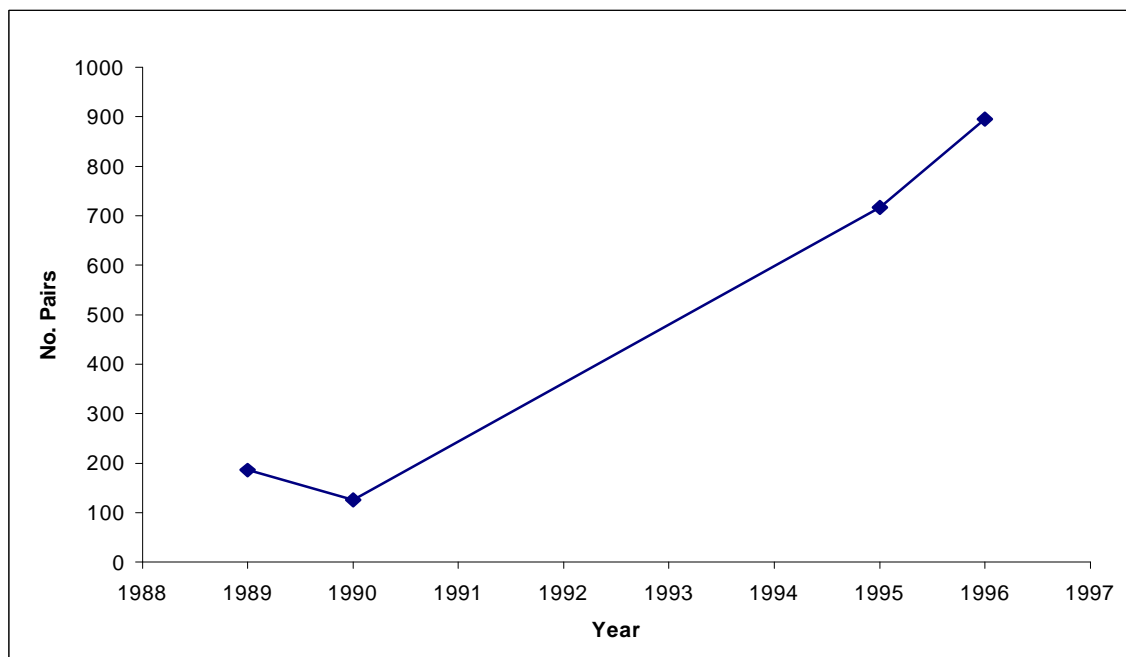
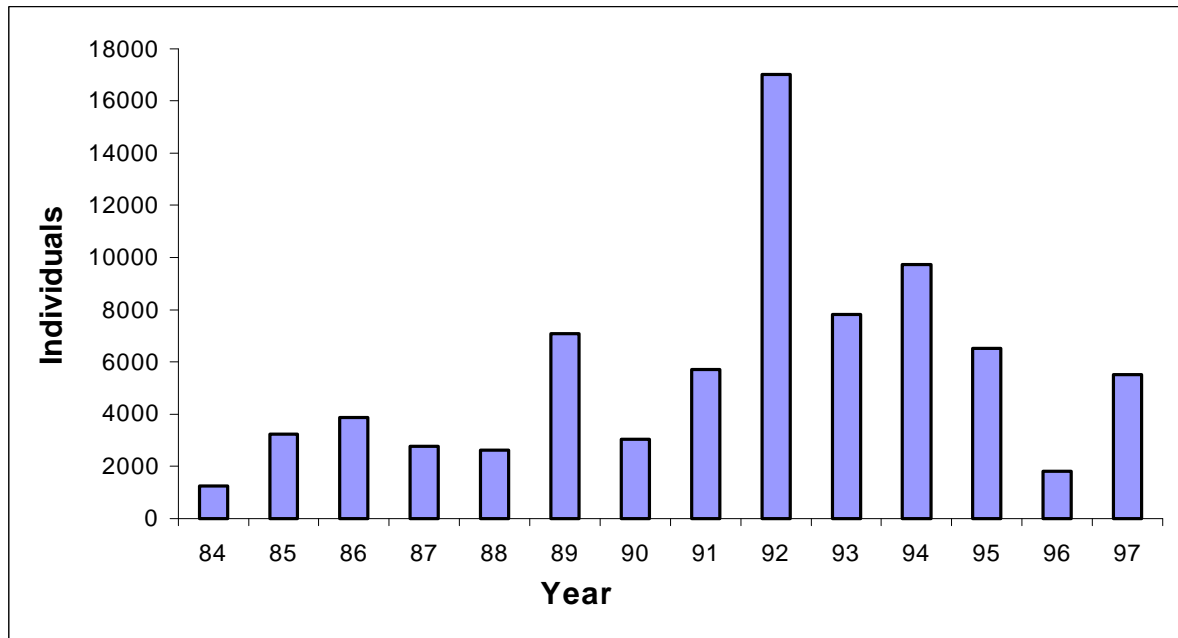


Figure 62. December estimates of DCCOs in South Carolina based on CBCs, 1984 – 1997.

Tennessee

Summary of Population Data and Trends

(Survey completed by R.M. Hatcher, Tennessee Wildlife Resources Agency, Nashville, TN)

Breeding.

The Double-crested Cormorant has recently returned to Tennessee as a nesting species, with the first nesting since 1955 documented in 1992 on an island in the Holston River inside a U.S. Army ammunition plant near Kingsport (Alsop, in press; Robinson 1990). Currently there are three known breeding sites, two in the eastern portion of the state, and one in the western portion in Humphreys county. Only the Holston colony near Kingsport in the northeast was active in 1999; at this time, 29 pairs were counted. This is the largest of the colonies and has increased since initial nests were discovered in the mid 1990s (Figure 63) (B. Hatcher, pers. comm.).

Winter.

A small number of cormorants winter at Reelfoot Lake and a few other sites within the state. Robinson (1990) reported that in recent years, winter flocks have generally numbered fewer than 100 birds. However, Jackson and Jackson (1995) show increases to about 250 birds between 1990 – 1992 at Reelfoot Lake, and note that Christmas Bird Counts from the late 1970s

to the present indicate numbers at Reelfoot Lake exceeded those of the 1940s. During the 1992 Christmas Bird Count an unusually high count of 661 birds was recorded at this location (LeBaron 1992). Nevertheless, winter numbers at Reelfoot Lake have not come anywhere near the high count of 4,500 reported for this location on Dec 20, 1952 (Jackson and Jackson 1995), and since 1992 have ranged between only 3 – 260 birds. Between 1985 – 1998 numbers counted for the entire state have remained under 1000 during CBCs (Sauer et al. 1996), but numbers do appear to recently be increasing, though CBC data have not yet been analyzed for this period (Figure 64). After mid-winter, fewer birds are found within the state (Robinson 1990).

Migration.

Peak spring migrant numbers are recorded in April – May; in fall, peak numbers are observed in October (Robinson 1990). Numbers during migration are estimated to be around 2000 birds (B. Hatcher, pers. comm.). Flocks of 1000 – 1500 are reported on Mud Lake, Shelby County, while smaller flocks of approximately 100 birds have been observed on the Mississippi River near Reelfoot Lake (Alsop, in press). Migrants are also observed on the Mississippi River near Memphis (B. Hatcher, pers. comm.).

Historical Information

Alsop (in press) reported that the Double-crested Cormorant was abundant at Reelfoot Lake in the late 1800s, and existed there in the thousands into the late 1930s. The species was a permanent year-round resident at this site, and numbers “swelled” with the arrival of northern migrants in the fall (Jackson and Jackson 1995). Breeding was first documented in the state at this site in 1919; about 75 nests were counted. In 1932, about 200 nests were counted (Jackson and Jackson 1995); in the late 1930s, between 250 – 400 nests were estimated (Robinson 1990; Alsop in press). Breeding was documented at Reelfoot Lake through the early 1940s. However, in the late 1930s and early 1940s, several visitors to this colony reported numbers of cormorants shot; at this time, cormorants were shot regularly by locals for fish bait and for sport (Jackson and Jackson 1995; Alsop in press). Other colonies documented in the state include one at the Duck River Unit of the Tennessee National Wildlife Refuge, Humphreys County, which in April, 1949, had about 100 nests; and one at Kentucky Lake, Benton County, with about 10 nests in 1955. Robinson (1990) reports this latter nesting record as the last for the state, prior to that recorded in 1992. In 1963, the Refuge Manager at Reelfoot Lake reported that the cormorant population there was declining yearly; only one pair was seen in that year. Therefore, it is not clear when breeding at Reelfoot Lake ceased (Jackson and Jackson 1995).

The species formerly wintered in large numbers at Reelfoot Lake, and also at the Duck River Unit. At the latter site, 3000 were reported on 3 January 1954 (Robinson 1990). Alsop (in press) noted that indiscriminate shooting and probably the sudden increase of pesticides ended the days of the cormorant as a permanent resident in w. Tennessee. As numbers did elsewhere, the cormorant population in Tennessee plummeted in the 1960s and 1970s, and the species was initially listed as “In Need of Management” in 1976. In the 1980s, however, the species began to recover, and in the 1990s, breeding was documented in both west and east Tennessee. Additionally, more cormorants are being recorded annually on bird counts throughout the year.

Land Ownership

No information available.

Productivity

No estimates available.

Figure 63. Changes in number of breeding pairs in Tennessee, 1992 – 1999.

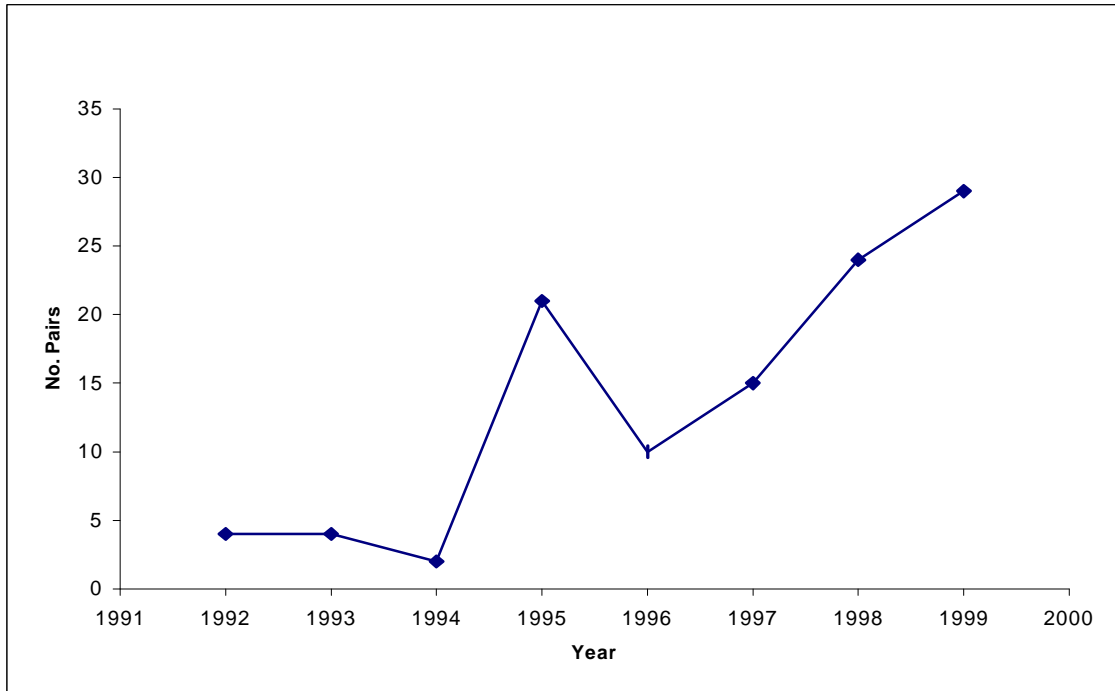
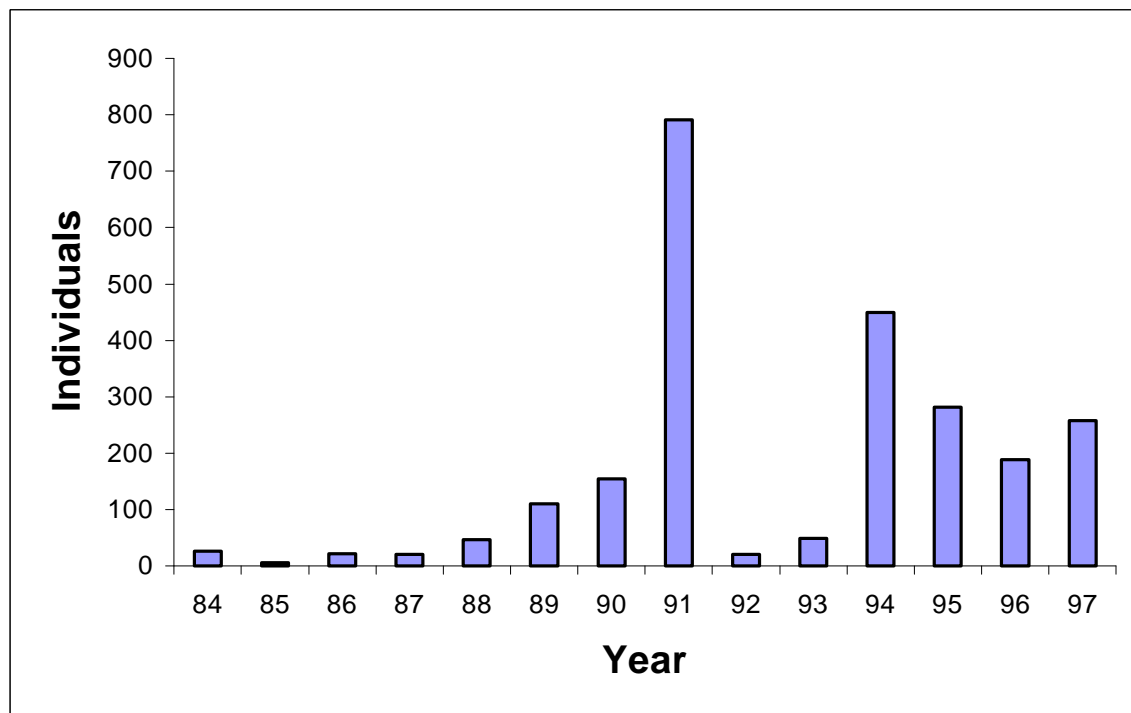


Figure 64. December estimates of DCCOs in Tennessee based on CBCs, 1984 – 1997.





Texas

Summary of Population Data and Trends

(Survey completed by J. Herron, Texas Parks and Wildlife, Program Director Wildlife Diversity; breeding data provided by B. Howe, USFWS, Albuquerque, NM.)

Breeding.

The Double-crested Cormorant was first documented nesting in Texas in 1926 (Oberholser 1974). In the 1990s, the species was found nesting in small numbers (≤ 50 nests per site) in the western and eastern portions of the state in Hansford, Hudspeth, Randall, Reeves, Sabine and Wood counties (B. Howe and R. Telfair II, pers. comm.). The species may also have nested in Delta, Harrison and Sabine counties (Texas Ornithological Society, 1995; B. Howe, pers. comm.). The quality of these records varies; in some of the northeastern counties (Harrison, Delta and Wood), the species may have been confused with the Neotropic Cormorant (R. Telfair II, pers. comm.). Better documentation of nesting records is necessary to accurately describe the distribution in Texas. At this time, population trends cannot be described; systematic surveys have not been conducted and very limited information is available on present numbers and distribution of breeding birds (J. Herron, pers. comm.).

Winter.

As in other southern states, cormorants are increasing in Texas; increases there have been described as “tremendous” over the last 30 years (J. Herron, pers. comm.). Christmas Bird Counts conducted between 1959 – 1988 suggested an increase of 14.5 % per year. More recent Christmas Bird Counts (1988 – 1998) estimated approximately 50,000 – 90,000 birds wintering in the state, and suggest that numbers are still increasing, but these data have not yet been analyzed (Figure 65). High counts between 83,000 – 90,000 were made 1992 – 1995. During these years, Christmas Bird Counts were conducted at Calaveras and Lake Livingston (lower), but these sites, which had very large numbers of cormorants, were dropped from the count beginning in 1995. However, numbers remained high in 1995 partly because an unusually high count of nearly 26,000 birds was counted in that year at Texarkana; the high at this site during the last decade had previously been around 13,000 (Sauer et al. 1996).

Very large numbers winter at reservoirs east of Dallas / Ft. Worth. Wintering populations are found on over 30 reservoirs, and resident populations are found on hundreds of smaller reservoirs or impoundments (J. Herron, pers. comm.).

Migration.

It is assumed that areas in the state also provide important stopover sites for migrating DCCOs. However, it is not possible to identify peak times or concentrations, as it is very difficult to distinguish wintering from migrating birds (J. Herron, pers. comm.).

Related Species.

See Related Species. in Louisiana profile (p. 122) regarding importance of distinguishing between Neotropic and Double-crested Cormorants in states where Depredation Order is in effect.

Historical Information

Audubon (1843) reported that the Florida Cormorant (*P.a. floridanus*) was “constantly resident in the Floridas and their Keys, and along the coast to Texas,” and that the species was abundant. Oberholser (1974) notes a record of the species in Tom Green County obtained in the fall of 1880. The earliest documented breeding record we obtained for the state was in 1926 at Matagorda Peninsula. Nesting was also documented in the 1930s, at which time the Double-crested Cormorant was a rare and local breeder. After 1939, breeding appeared to cease for more than 30 years, with the last nesting record obtained in Wilbarger County. The Cooperative Fish-eating Bird Survey, conducted in late May and early June, 1969, along the entire coast found no Double-crested Cormorants (Oberholser 1974). Nesting was not documented again until the mid 1970s, when 14 nests were found on the Toldeo Bend Reservoir at the mouth of Mill Creek Bay in April 1974 (Holm et al. 1978); five nests were also reported in Midland County in 1973, but the outcome of these nests was not clear (B. Howe, pers. comm.). Breeding has since expanded elsewhere within the state.

Oberholser (1974) noted that this species was still numerous and widespread in Texas as a migrant and winter resident. He described it in winter (mid-November to early-March) as “locally very common to uncommon along entire coast; locally fairly common to uncommon inland in eastern half; scarce to rare west of 100th meridian.” In recent years numbers have increased dramatically; very large numbers winter in reservoirs east of Dallas / Ft. Worth. Oberholser noted that the many hundreds of livestock watering lakes constructed in the early 1940s – early 1970s aided it. Additionally, J. Herron (pers. comm.) stated that, while many factors have likely contributed to the species’ growth, the hundreds of artificial lakes and reservoirs created new habitat that this species was able to readily occupy.

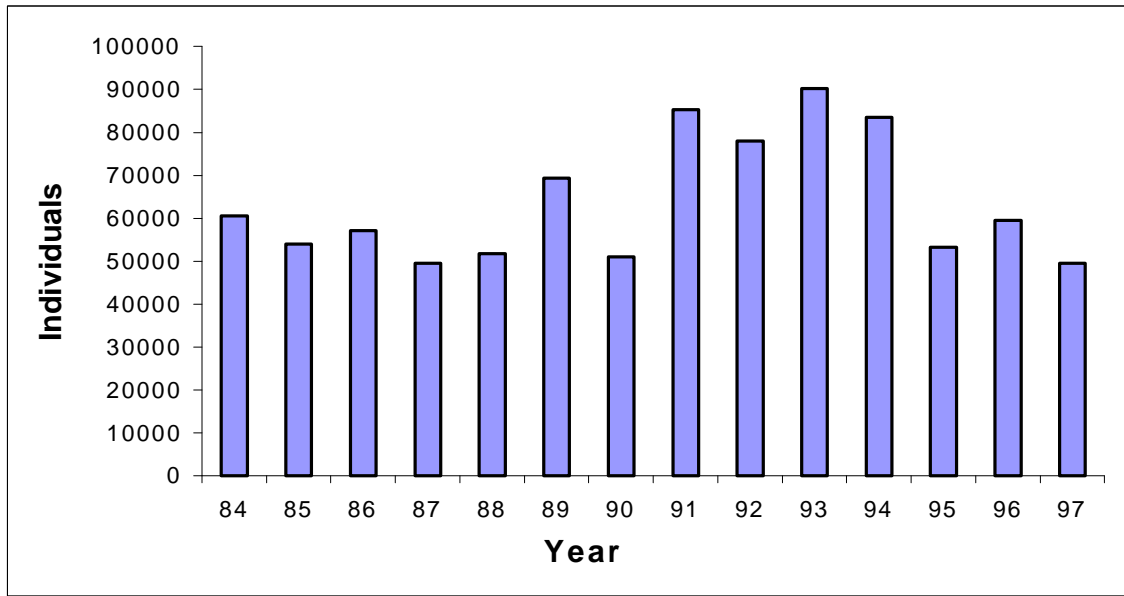
Land Ownership

No information provided.

Productivity

No estimates provided.

Figure 65. December estimates of DCCOs in Texas based on CBCs, 1984 – 1997.



ZONE 5: SUMMARY OF POPULATION DATA FOR THE NORTHEAST ATLANTIC COAST

Introduction

Breeding Information

In this zone, the breeding population extends along the Atlantic coast from s. Newfoundland, the north shore of the Gulf and estuary of the St. Lawrence River, Anticosti Island, Magdalen Island., south to New York City and Long Island. Table 6 shows the status (Breeding, Wintering, Migrant) of the Double-crested Cormorant in this region. Over the last two decades the number of breeding Double-crested Cormorants has increased along much of the northeast Atlantic Coast; currently the breeding population is roughly estimated at about 86,000 pairs. Of 15 states and provinces for which we had data on breeding cormorants, eight had increasing cormorant numbers, one had declining numbers, one had possibly declining or stable numbers, one had possibly increasing or stable numbers, and trends in four could not be determined.

Eastern Canada.

Very large numbers breed in Québec and the surrounding area, including the St. Lawrence River and Estuary, the North Shore, and multiple islands (see Québec summary). Colonies in southwestern Québec were grouped with the interior population, based on Hatch and Weseloh (1999), and on apparent spatial separation from other Québec locations and proximity to interior colonies when viewing colony distribution. Québec has the largest number of breeding cormorants in the Northeast Atlantic. Numbers have increased substantially in Québec since 1970, especially in the St. Lawrence River. These increases may be related to reduction in human persecution and changes in prey fish communities (Chapdelaine and Bédard 1995). In 1989, a large scale culling program was introduced and reversed population trends in the St. Lawrence River Estuary, where rapid growth was occurring.

Very large breeding concentrations are also found on Nova Scotia and Prince Edward Island, where rapid increases began occurring in the 1970s. However, during the last decade declines have occurred on these two provinces, which may indicate that numbers are beginning to stabilize (see summaries). In New Brunswick, cormorants were increasing as of 1997, but the current rate of increase has slowed (Korfanty et al. 1997). In Newfoundland, the number of breeding birds is comparatively very low, and trend information is not available.

New England and U.S. Coast.

Very large breeding concentrations occur in New England along the coasts of Maine and Massachusetts. With the exception of Maine, (where numbers began declining between the mid-1980s and early 1990s, and where numbers may have reached carrying capacity), rapid increases have occurred since the 1970s. In parts of southern New England (Connecticut, Rhode Island, coastal New York) the cormorant has been documented fairly recently as a breeding species; mostly steady increases have occurred in these states, and numbers are growing fairly rapidly. However, due to declines in Maine, numbers in New England have declined overall between 1992 (and perhaps earlier) and 1995 at an average annual rate of - 6 % (Figure 66). Further south along the Atlantic, the species has expanded to or possibly re-colonized (see Historical

Information, below) several states, with first breeding records obtained in New Jersey, Pennsylvania, Maryland, Delaware and Virginia between the late 1970s and 1990s.

Winter and Migrant Birds

Small numbers (< 1000 birds) winter in some of the New England states, and several thousand likely occur in Virginia and coastal New York. CBC data collected between 1984 – 1998 (Sauer et al. 1996) suggest that numbers may be increasing (Figure 67), particularly in Virginia, but data have not yet been analyzed; a few more years of data may be necessary to document current trends. Very large numbers of fall migrants are observed along the coast, with hundreds of thousands reported from southern New England to Virginia.

Historical Information

Breeding has been documented for this zone for nearly 500 years; Hatch (1982) reported that bones of adult and young cormorants found in Indian middens in Boston Harbor, Massachusetts, dated back to as early as 1500. Numerous early records also exist for New Brunswick, Newfoundland, Nova Scotia, Québec, Maine, Virginia, and possibly Rhode Island. Early records indicate that cormorants were very abundant in Atlantic Canada, New England and Virginia prior to European settlement. While the full extent of the historic distribution is not known, changes in numbers have been mostly within the known historic range (Hatch and Weseloh 1999). Sometime during settlement cormorants began declining in most parts of their Atlantic range. By the early 19th century they were entirely extirpated as breeders in New England. By the late 19th and early 20th centuries, cormorants were basically extirpated from Nova Scotia and greatly reduced in the other Maritime Provinces and Québec. Declines across the range appear to have been due to human persecution and exploitation. Numerous authors have suggested that, given pre-settlement habitat, the known historic distribution of cormorants, and other evidence, cormorants may have been present in areas for which we have unclear or no early (pre-20th century) records, such as New York, Rhode Island, and Virginia (Mendall 1936; Arbib 1998; Ferren and Myers 1998). Thus, some of the areas “colonized” in the last twenty years may actually represent a re-colonization rather than a range expansion.

New England

(Summarized from Krohn et al. 1995, unless otherwise noted.) In New England, cormorant numbers began to recover around 1925, when they began consistently breeding and increasing in Maine. Birds may have immigrated from eastern Canada. The cormorant returned as a breeding species to southern New England in the late 1930s and early 1940s, when nesting was recorded in Massachusetts, and later in New Hampshire during the 1940s or 1950s (J. Kanter, pers. comm.). By the mid-1940s numbers in Maine were perceived to cause problems for fishermen and a cormorant control program, utilizing egg spraying, was initiated. The control program appeared to coincide with a reduction in population growth (Drury 1973); while growth in New England continued, it did so at a very low rate until the 1970s (Hatch 1982; Veit and Petersen 1993). Low growth rates may have been due to the combined effects of control efforts, pesticides and natural limitations to population size. After the DCCO received protection under the Migratory Bird Treaty Act and pesticide regulations were introduced, a second period of growth began in the 1970s and has continued through the 1990s, though numbers in Maine have begun to decline and are possibly stabilizing (R.B. Allen, pers. comm.). From the late 1970s through the 1990s, growth rates have been greater in southern New England than in Maine,

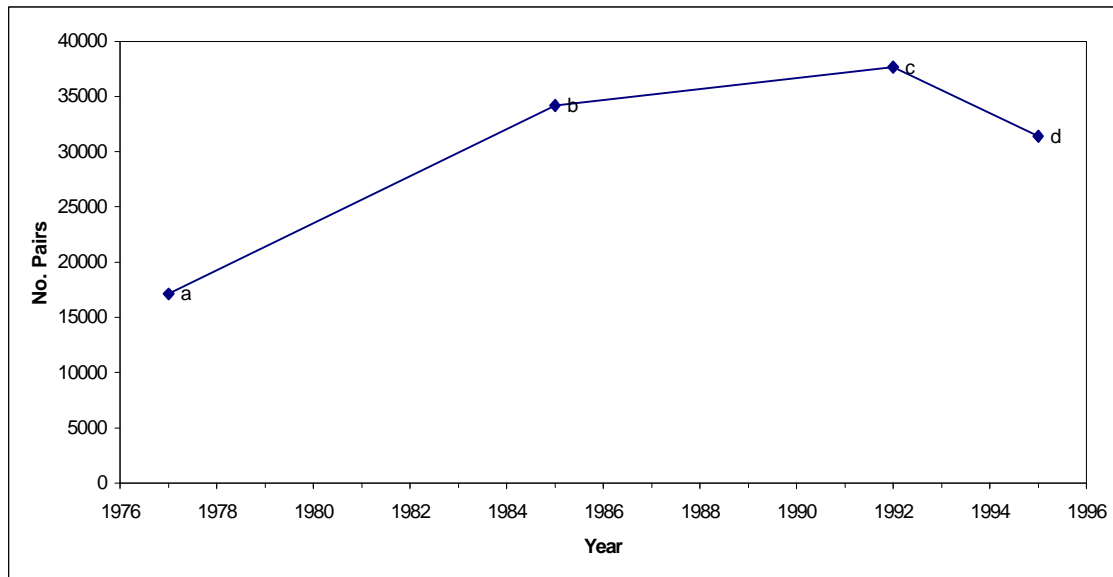
though Maine still supports the majority of cormorants breeding in New England. During the mid-1980s through the early 1990s growth rates in both areas decreased.

Eastern Canada.

We were not able to obtain much detailed historic information on cormorants in Newfoundland, New Brunswick, or Prince Edward Island. However, the limited information we did obtain suggests that the history of cormorants in eastern Canada was similar to that in New England, though numbers in Canada did not appear to drop as low as those in New England. In the Maritime Provinces, cormorants were reduced to remnant numbers by 1900 (Erskine 1992). Lewis (1929) notes early records of several colonies in the Québec region (including the north shore of the Gulf of St. Lawrence, and the Gaspé peninsula), Newfoundland, Nova Scotia and New Brunswick, that had all disappeared by the time he was writing or earlier in this century. He also provides an estimate of the number of breeding birds in these provinces, roughly calculated around the time he was writing, and there appears to have been about 5000 pairs.

Lewis's estimate (1929) is the only one for all of eastern Canada prior to recent estimates. During the 1940s and 1950s, when cormorant numbers were known to be increasing in New England, they also increased in the Québec region and in Nova Scotia, while numbers were low on Prince Edward Island. No information is available for New Brunswick and Newfoundland at this time. In the 1970s, numbers began rapidly increasing in Québec, Nova Scotia and Prince Edward Island.

Figure 66. Changes in numbers of breeding pairs in New England, 1977 – 1994/95.



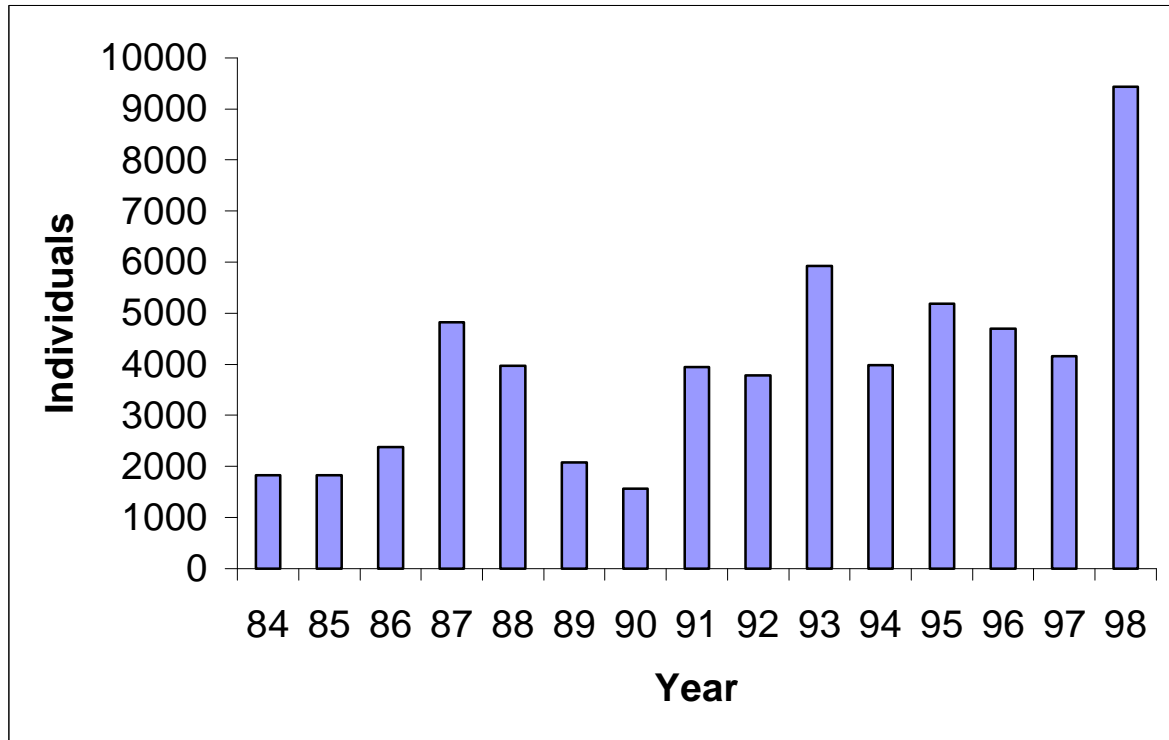
a = 17,117 pairs, 1977, included ME, NH, MA (Krohn et al. 1995).

b = 34,170 pairs, mid 1980s, includes ME, NH, MA, RI, CT (Krohn et al. 1995).

c = 37,648 pairs, early 1990s, includes ME, NH, MA, RI, CT; numbers for ME based on visits to 67 colonies, about half the number of known colonies (Krohn et al. 1995).

d = 31,391 pairs, 1994-95, includes ME (numbers declined substantially since mid 1980s), MA, RI, CT, coastal NY (this report).

Figure 67. December estimates of DCCOs in the Northeast Atlantic, Zone 5*, based on CBCs, 1984 – 1998.



* States included: CT, DE, MD, NJ, NY, PA, VA; annual totals based on sites with ≥ 100 individuals.

Table 6. Zone 5, Northeast Atlantic Coast. Summary of Population Data and Trends
(+ = increasing; - = decreasing; 0 = no change).

State/Province	Status (B, M, W, w) ¹	Number colony sites/ time period	No. active col./ last yr. surveyed	No. pairs/ last year surveyed	Breed. Trend	Winter number (Birds)	Migrant number (Birds)
CT	B, w	26/1986-98	20/1998	961	+	25-100	-
DE	B, M, w	1/1991	1/1991	3	NA	NA	1000s (spring); 100,000s (fall)
ME	B, M	≥ 117/1980s-90s	117/1994-95	17,100	-	-	1000s (spring)
MD	B, M, w	4/1990s	4/1999	1046	+	-	100,000s (fall)
MA	B, M, w	28/1977-95	27/1994-95	7833	+	200	NA
NB ³	B, w (?)	15-25 ³ /1986-91	15/1986-91	9,278	+	NA	-
NF	B	≥ 7/1980-96	≥ 5/1996	343	NA	-	-
NH	B, M	3/1974-98	1/1998	20	NA	-	NA
NJ	B, M, w	5/1993-97	5/1993-97	NA	NA	200-1000	189,768 (fall)
NY (coast)	B, M, W	6/1995	6/1995	3,528	+	1000s	1000s
NS ⁴	B	71-80 ⁴ /1987-92	69/1987 ?/1992	15,964 12,000	-/0 (?)	-	-
PA	B, M	1/1996-99	1/1999	5	NA	-	100s-1000s
PEI	B, M (?)	12/1978-99	8/1999	7,695	+/0 (?)	-	1000s (?)
QC	B	99 ⁵ /1970-98	52 ⁶ /1987-93	>24,128 ⁷ (1987-93)	+	-	-
RI	B, M, w	10/1980-98	9/1997	2056	+	200	10,000-100,000
VA	B, M, W	1/1993-97	1/1997	60-70	+	1000s	20,000-300,000
WV	M, w	NA	NA	NA	NA	<100?	1000s
Totals		≥ 406	≥ 341	> 86,056		NA	NA

1 = B, Breeding; M, Migrant; W, Wintering; w, small numbers winter.

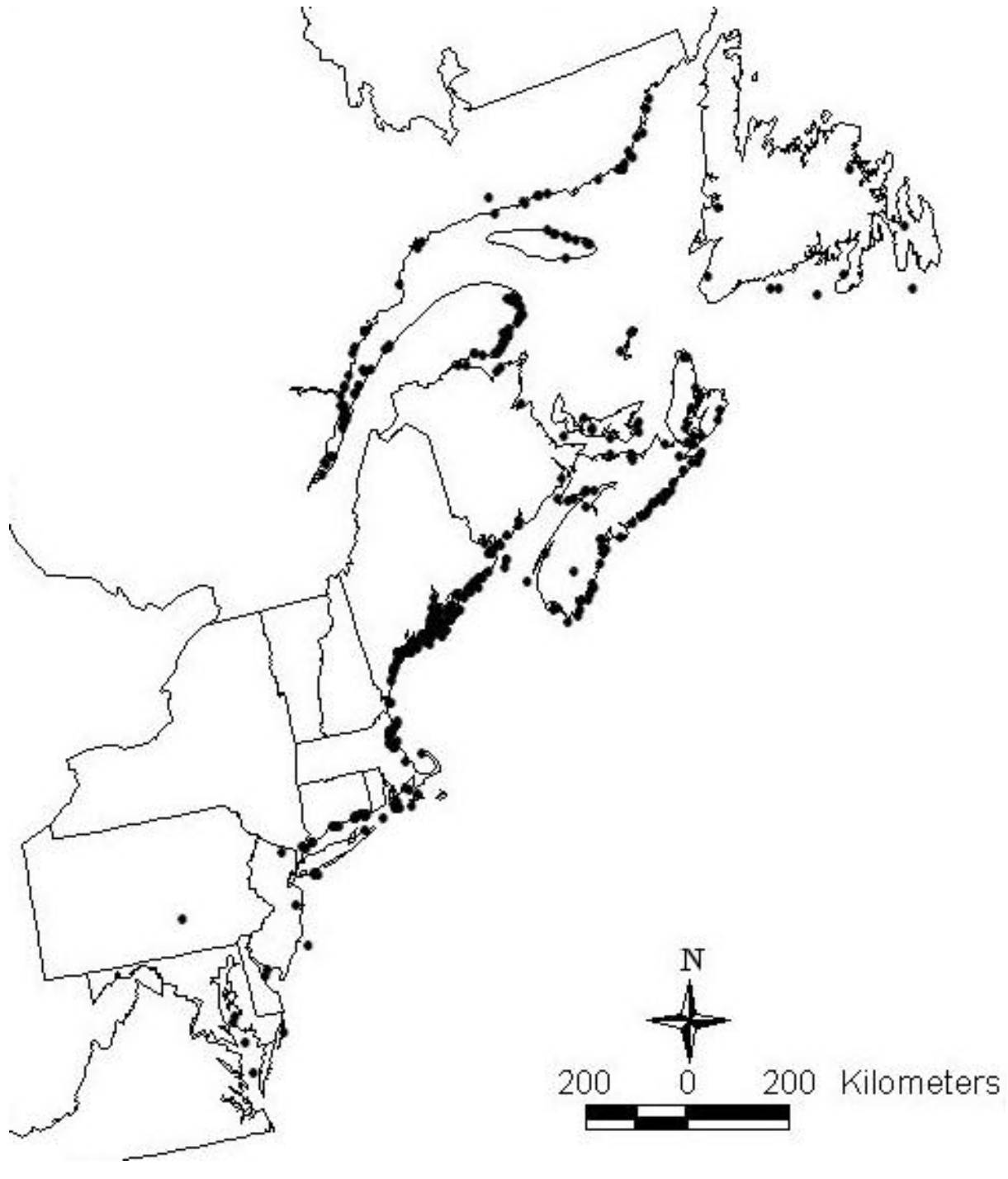
2 = Counts for 5 Long Island colonies not available this year, so 1995 census data used for these colonies. Estimate is probably within 10 % of actual population size (B. Miller, pers. comm.).

3 = Numbers from Lock et al. (1994); Korfanty et al. (1997) reported approximately 15,000 individuals in summer months, but did not provide breeding number or information on how estimate was obtained or time frame. Erskine (1992) reported 25 colonies in “recent” surveys by provincial and federal officials, but no other information was provided.

4 = Lock et al (1994) provided number of active colonies in 1987; Erskine (1992) reported 80 colonies in “recent” surveys by provincial and federal officials, but no other information was provided. Korfanty et al. (1997) provided estimate of breeding pairs in 1992, but did not report information on number of active colonies.

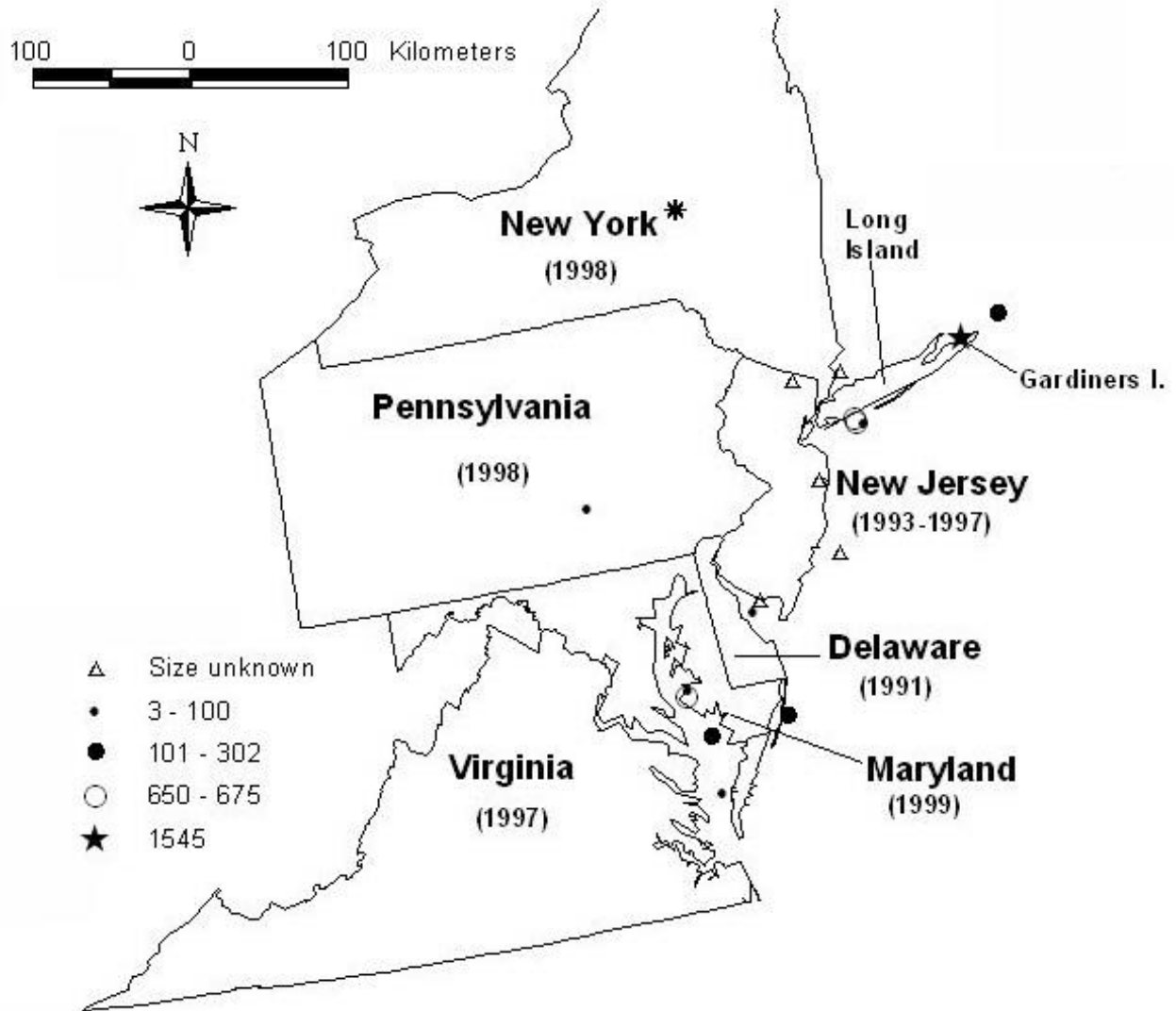
- 5 = A total of 121 known colonies was reported for Québec, but 22 of them appeared to have disappeared during the nearly 30 year period for which data were supplied. We included only those colonies that were active in the last year they were visited (J.F. Rail, pers. comm.).
- 6 = This is the number of colonies with breeding birds that were visited around 1990 (1987 – 1993). Many colonies were not visited during this time period, but were and are currently presumed to be active (J.F. Rail, pers. comm.).
- 7 = No province-wide surveys have been conducted during a single year in Québec. Number of pairs is based on counts conducted around 1990 (all colonies surveyed between 1987 – 1993).

Figure 68. Distribution of Northeast Atlantic Coast DCCO breeding population (Zone 5), 1980 – 1999¹



¹ A small number of colonies in Quebec have not been surveyed since the 1970s, but are presumed active (J.F. Rail, pers. comm).

Figure 69. Distribution and size of active colonies in the mid-Atlantic states at time of last complete surveys.*



* Distribution and size of inland New York colonies shown in Figure 35.

Figure 70. Distribution and size of active colonies in New England at time of last surveys.

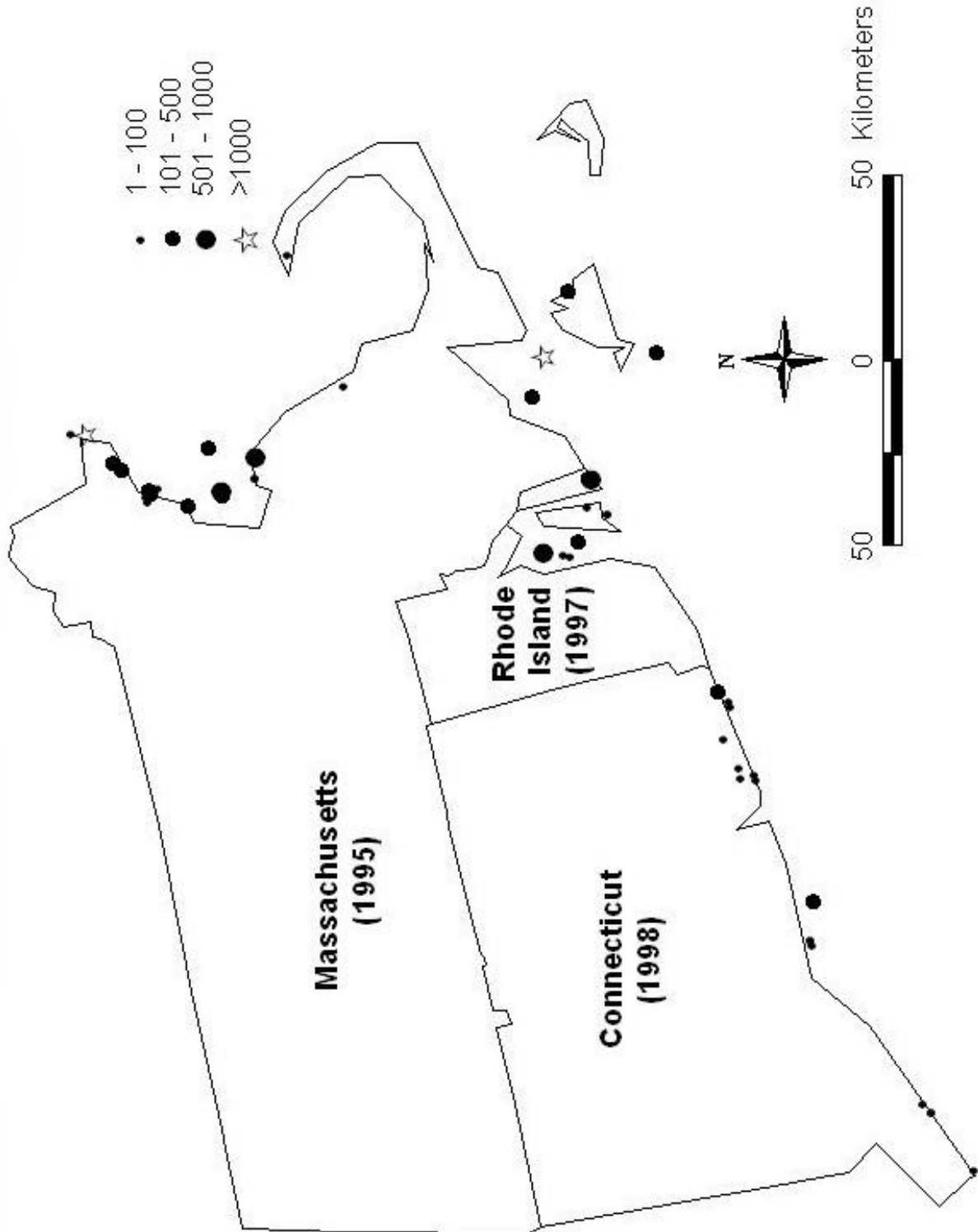


Figure 71. Distribution and size active colonies in Maine during last complete survey (1994 – 1995).

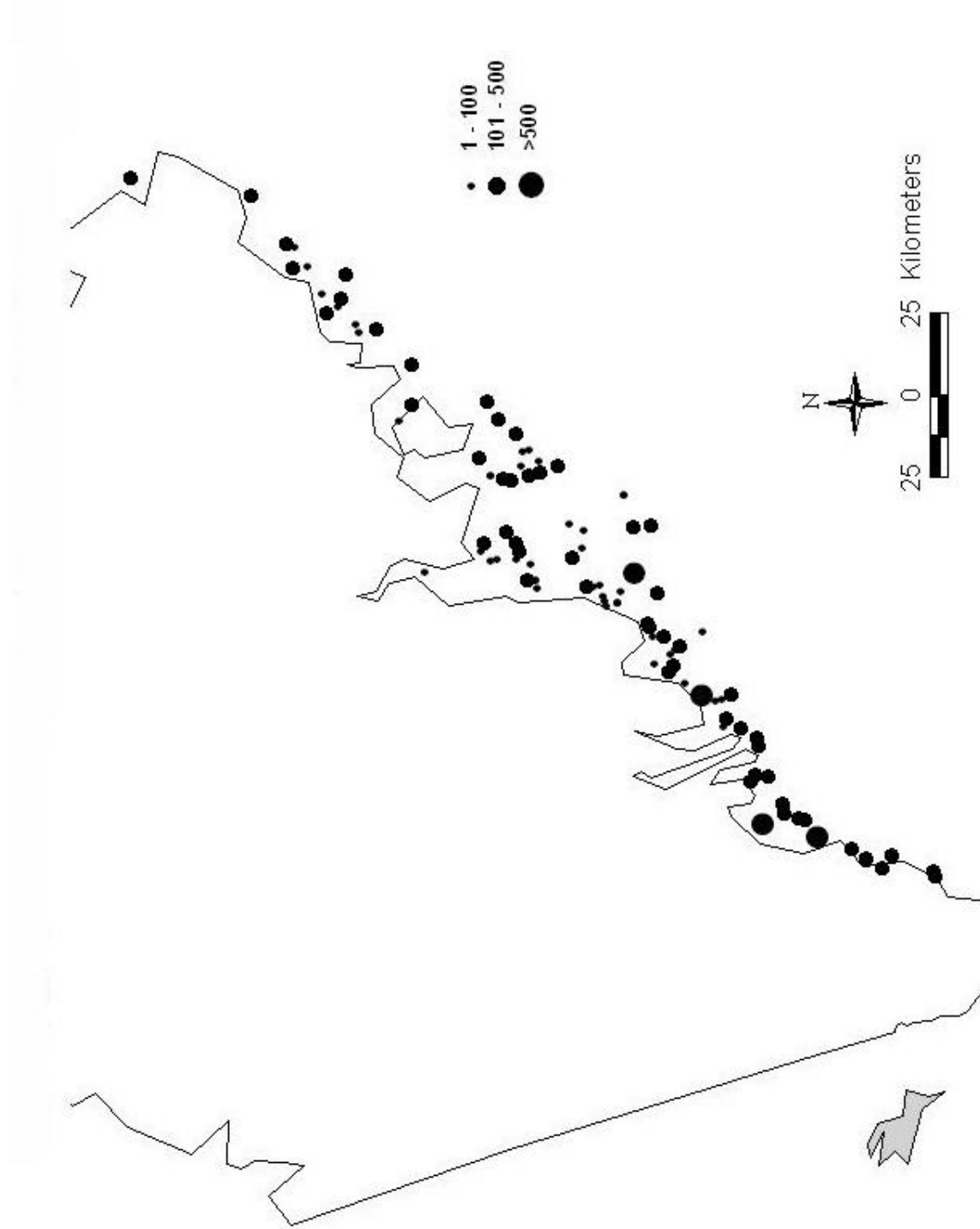
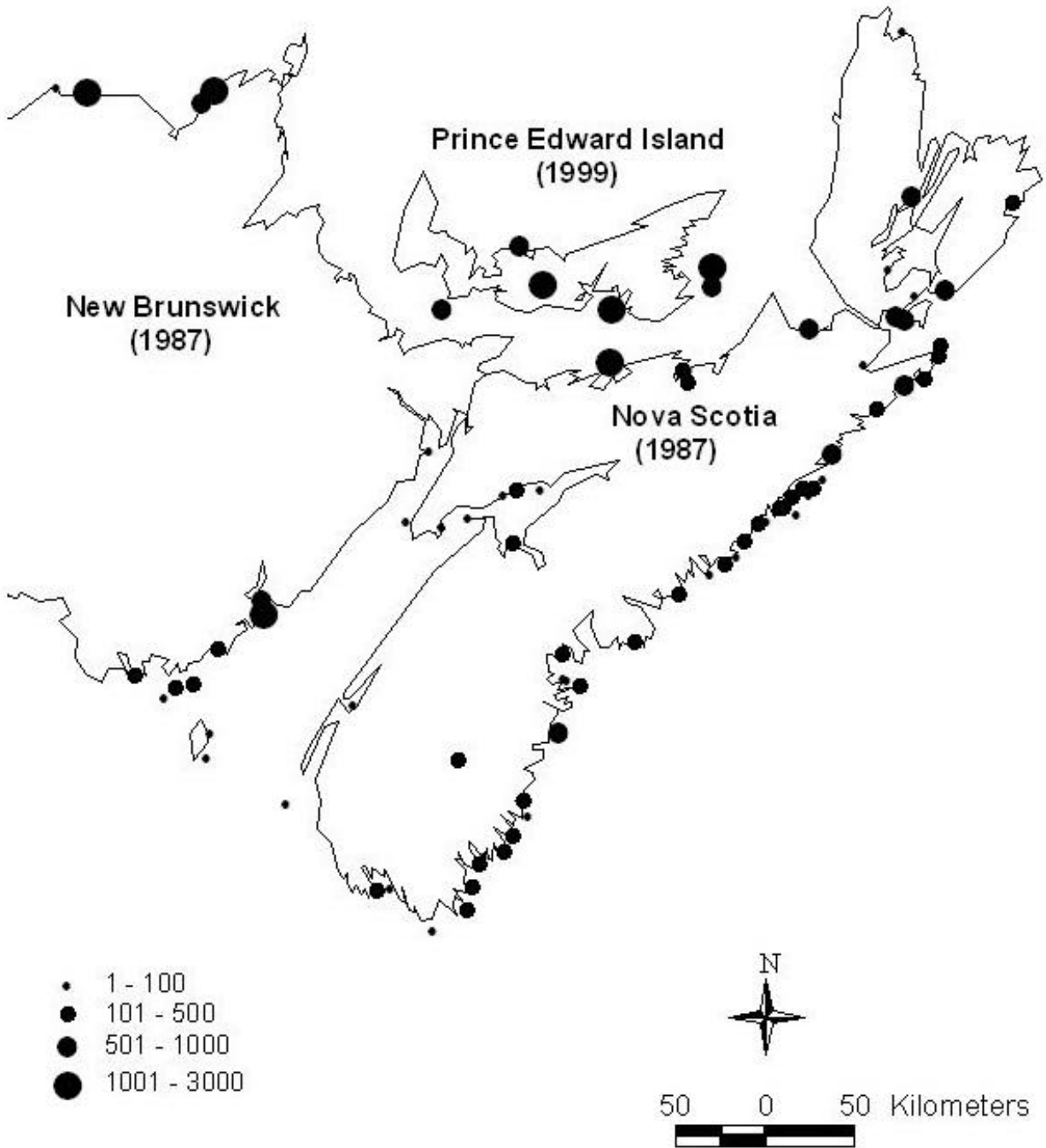


Figure 72. Distribution and size of active colonies in New Brunswick, Nova Scotia¹ and Prince Edward Island during last complete surveys.



¹ A complete survey was conducted for Nova Scotia in 1992, but we were not able to obtain this information; locations and sizes shown are for 1987.

Figure 73. Distribution of active colonies in Quebec at time of last surveys (1970 – 1998).

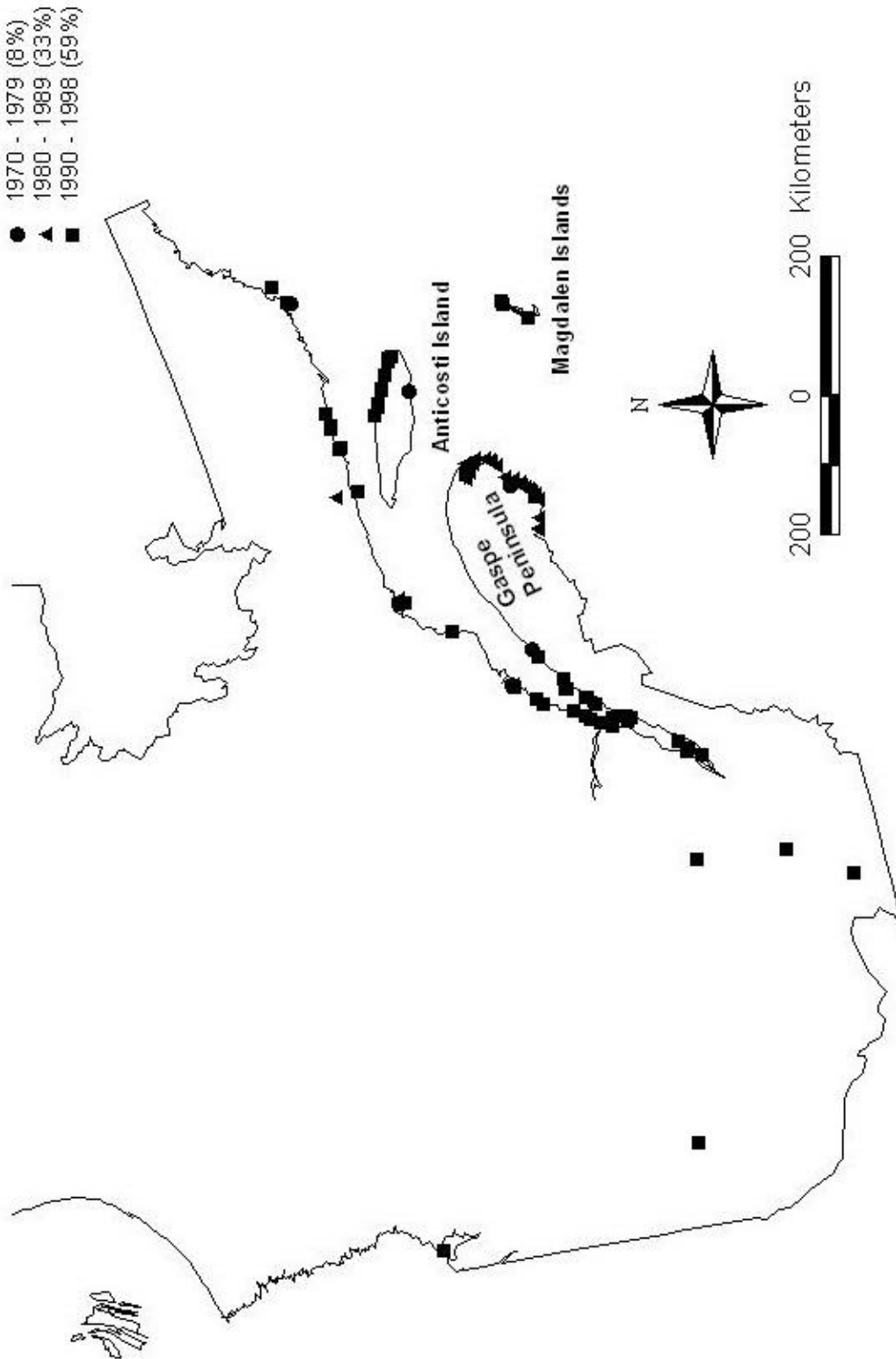
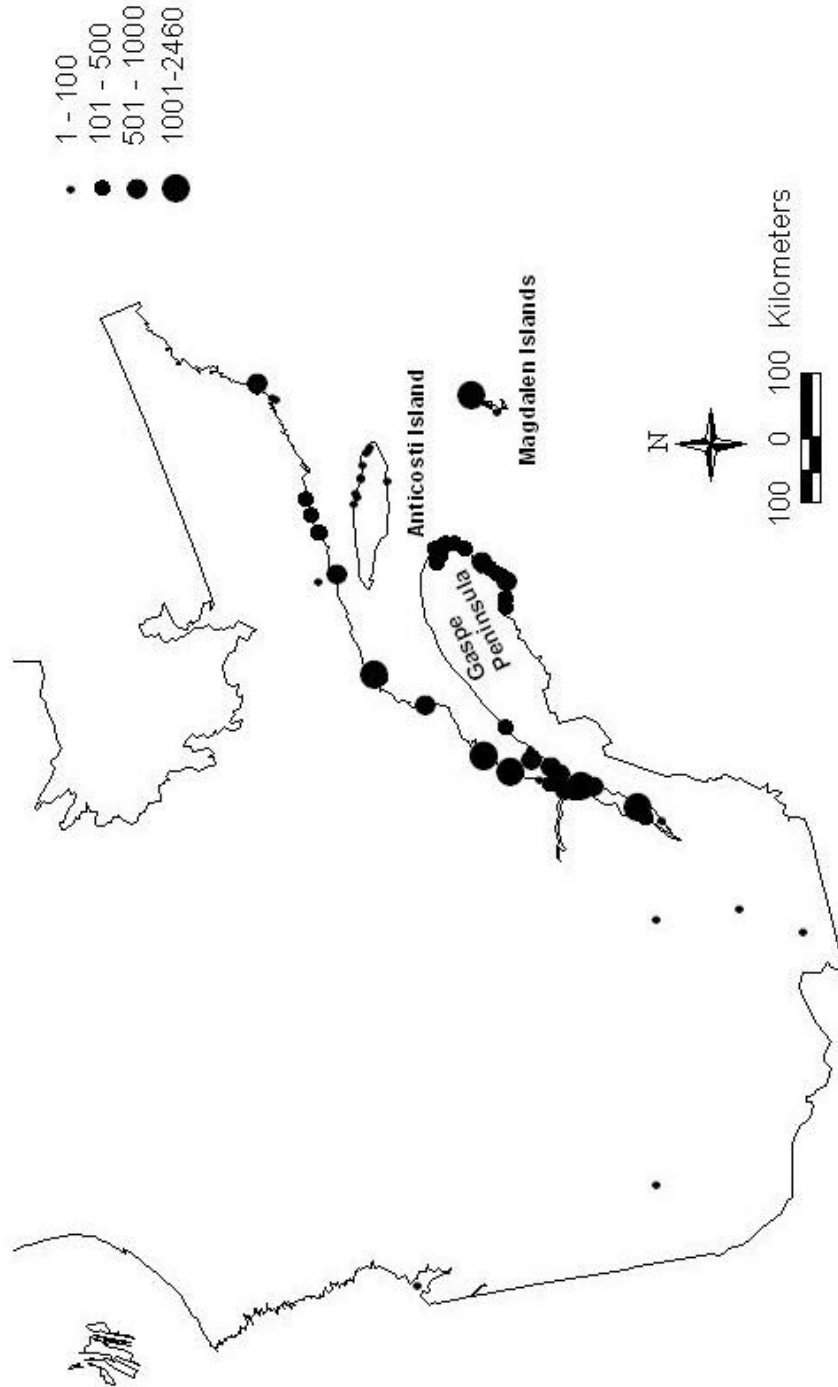
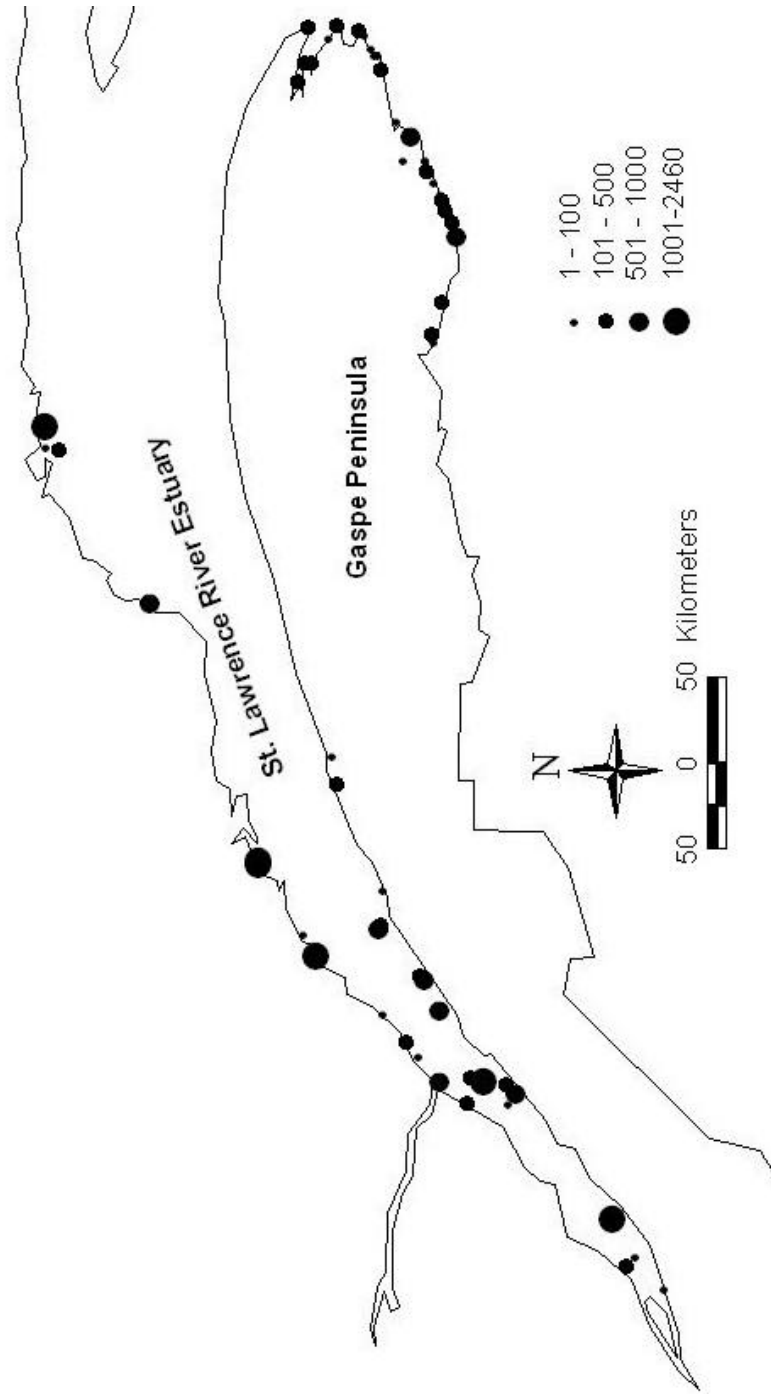


Figure 74. Distribution and size of active colonies in Quebec at time of last surveys (1970 – 1998).¹



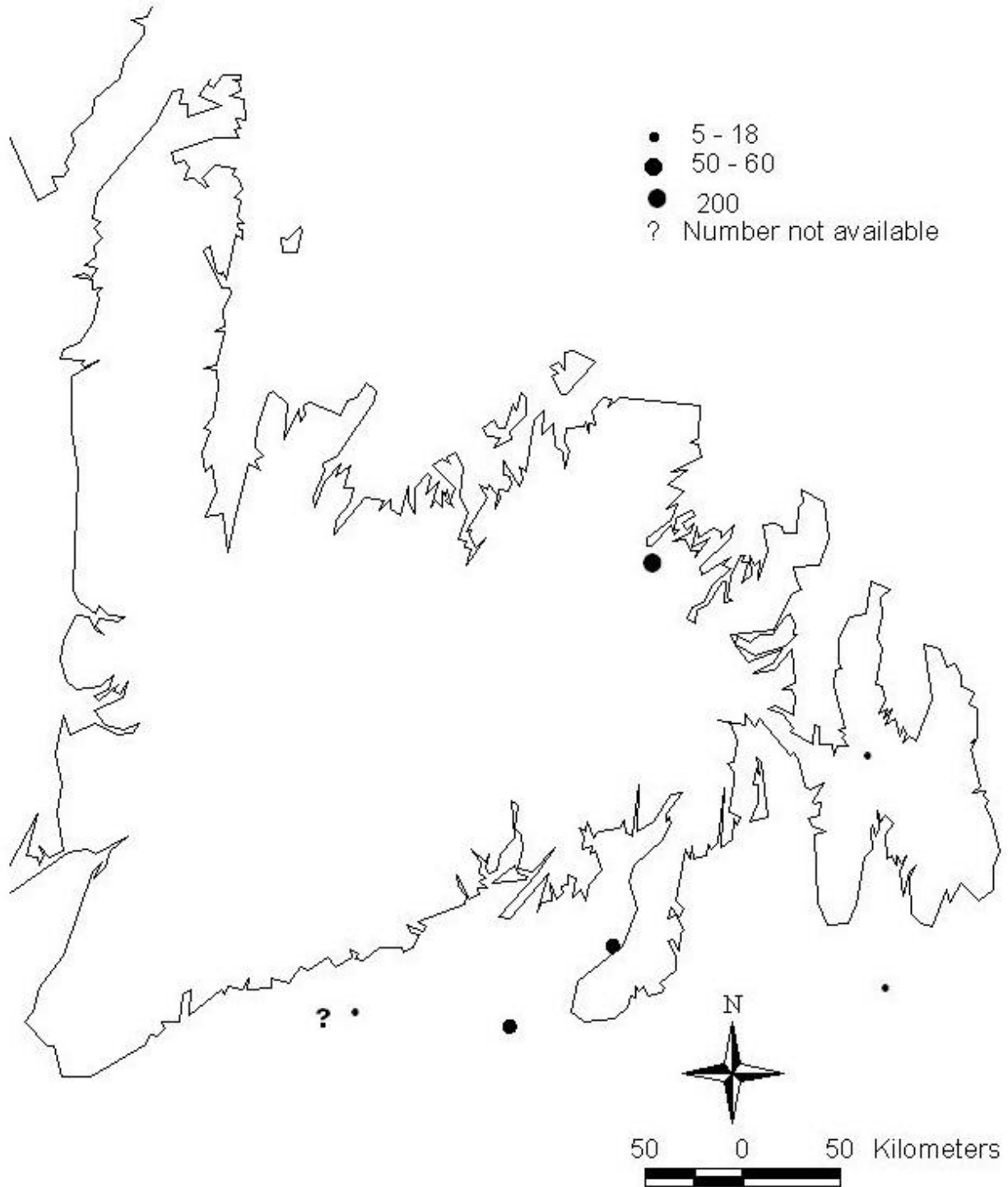
¹ All colonies shown are presumed active, though some have not been surveyed since the 1970s and 1980s (see Figure 73). Sizes shown are based on last colony visits (J.F. Rail, pers. comm.).

Figure 75. Detail of the St. Lawrence River Estuary and Gaspé Peninsula showing distribution and size of active colonies at time of last surveys (1970 – 1998).¹



¹ All colonies are presumed active, though some have not been surveyed since the 1970s and 1980s. Sizes shown are based on last colony visits (J.F. Rail, pers. comm.).

Figure 76. Distribution and size of active colonies along the coast of Newfoundland and on offshore islands during last complete survey (1996).



Connecticut

Summary of Population Data and Trends

(Survey completed by J. Victoria, Dept. of Environmental Protection, Franklin Wildlife Management Area, North Franklin, CT.)

Breeding.

The Double-crested Cormorant appears to be a relatively recent breeder in Connecticut, with first nesting documented in 1979 at East White Rock in the Norwalk Islands (Wood 1979). Currently there is a total of 26 known colony sites; complete censuses have been conducted every three years since at least 1986, when 181 pairs were recorded. Between 1986 and 1989, numbers increased by more than 600 %, and reached a peak of 1117 pairs. But by 1992, numbers had declined by close to 50 %. Censuses in 1995 and 1998 indicated that numbers were again increasing (Figure 77); in 1998, the last survey year, a total of 961 pairs was estimated (J. Victoria, pers. comm.). Between 1986 – 1998, the average annual rate of increase was 15 %.

Winter.

Along with the increase in breeding birds, in the 1980s the species became regular in winter along the coast and inland during migration (Sibley 1994; Zeranski and Baptist 1990). However, only small numbers of birds, 25 – 100 individuals, are estimated to winter in the state (J. Victoria, pers. comm.). Review of CBC data collected between 1984 and 1998 identified one site, New London, where > 100 birds have been estimated during a count (Sauer et al. 1996).

Migration.

Although observations of migrants are routine, significant concentrations do not stopover for any length of time (J. Victoria, pers. comm.).

Historical Information

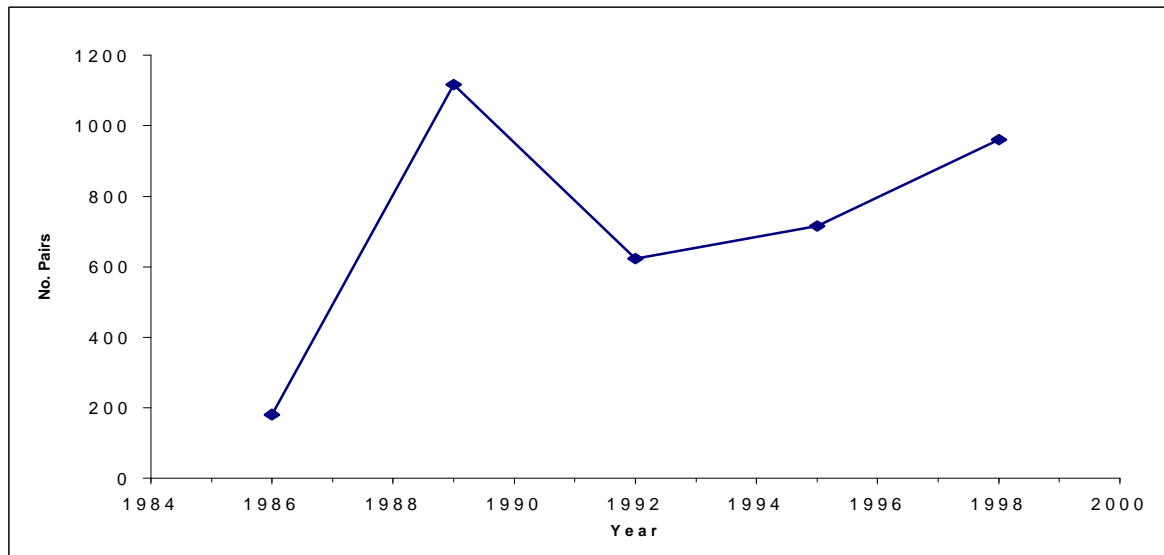
Sibley (1994) reports that no historical nesting records exist for the Double-crested Cormorant in the state of Connecticut, but that it was recorded as a rare migrant through the 1800s and into the early 1900s.

Land Ownership

No information available.

Productivity

No estimates available.

Figure 77. Changes in numbers of breeding pairs in Connecticut, 1986 – 1998.

Delaware

Summary of Population Data and Trends

(Survey completed by C.M. Heckscher, Division of Fish and Wildlife, Smyrna, DE.)

Breeding.

The Double-crested Cormorant appears to be a recent breeder in Delaware. Breeding was first documented in 1991 at Little River, where three pairs were counted. However, no other information on this site was available at the time this assessment was prepared (C. Heckscher pers comm.).

Winter.

Cormorants do winter in the state, but estimates of numbers and wintering locations were not provided. Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified one site, Rehoboth, where ≥ 100 birds were estimated during at least one count (600 birds in 1984).

Migration.

During spring migration, significant numbers are observed continuously along the coast, but no regular concentration sites were reported. Peak numbers in late April are estimated in the thousands; peak numbers migrating down the coast in the fall may be in the hundreds of thousands (D. Forsell, pers. comm.).

Historical Information

No information available.

Land Ownership

No information provided.

Productivity

No estimates available.

**Maine*****Summary of Population Data and Trends***

(Survey completed by R.B. Allen, Dept. of Inland Fisheries and Wildlife, Bangor ME.)

Breeding.

Maine, by far, has the largest number of breeding Double-crested Cormorants in New England. In 1994 – 1995, at least 17,100 pairs were estimated at 117 known colonies, all of which were on coastal islands, except for one inland nesting site (not shown on Figure 71; coordinates not provided). Numbers appear to have reached their peak in the mid 1980s, when 28,760 pairs were counted at 121 colony sites during 1984 – 1985 surveys. While Krohn et al. (1995) reported 28,000 pairs in the early 1990s, this estimate was an approximation based on visits to only about half of the known colonies. Numbers appear to have declined at an average annual rate of – 5 % during the last two complete surveys (1984/85 and 1994/95) (Figure 78). This decline may be a result of several factors: 1) in the mid-to-late 1980s, cormorants may have exceeded the carrying capacity for their numbers in coastal Maine; current numbers are probably more in line with what the habitat and fish resources can actually support; 2) Bald Eagles have increased in recent years in Maine and Bald Eagles have been observed to heavily prey on chicks and disrupt cormorant colonies; 3) aerial surveys show large variation depending on when birds are counted during the nesting cycle (R.B. Allen, pers. comm.).

Winter.

Cormorants are not known to winter in Maine (R.B. Allen, pers. comm.). Review of CBC data collected 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were estimated during at least one count, though the species was occasionally found in small numbers.

Migration.

During spring migration, peak numbers (thousands) are seen in April, migrating north to the St. Lawrence River and Gulf; no regular stopover sites reported (R.B. Allen, pers. comm.).

Historical Information

Mendall (1936) described the early history of the Double-crested Cormorant in Maine, and his work is summarized here, unless otherwise noted. The first mention of Cormorants in New England was made by Captain John Smith in 1616. In the summer of 1614, Captain John Smith explored the Maine coast in a southwesterly direction from the Penobscot River, and Mendall (1936) felt that his reference to “Cormorants” likely applied to Maine. Towards the end of the 17th century, reports of how the Indians captured cormorants around Black Point, Maine (now the town of Scarborough), were made, but it is not clear if the species in question was *P. auritus* or *P. carbo* (Mendall 1936). Confusion between these two species and over the status of cormorants is apparent in much of the literature that followed. However, by the 19th century, for reasons that are not clear, “the cormorant” was extirpated as a breeding species along the Maine coast (Gross 1944). At the end of the 19th century, Smith (1883, cited in Mendall 1936) wrote that no Double-crested Cormorants bred on the coast of Maine.

Though it was observed roosting, summering and migrating along the coast at various times, the Double-crested Cormorant was not documented as a breeder in Maine until the late 1800s. The first Double-crested Cormorant eggs collected in the state were probably taken in 1892 near Isle au Haut. The following year seven nests were observed, four of them with eggs, on Black Horse Ledge. Nesting was again observed at this site in 1895 and 1896. Egg collecting at these sites was believed to have contributed to the disappearance of nesting DCCOs in Maine in the 1890s (Gross 1944). Though many young and adult cormorants were observed loafing in several areas, the species was not recorded nesting in Maine again until 1925. The population then increased rapidly through the mid 1940s. In 1931, more than 1700 individual breeding birds were observed at various locations. By 1944, Gross (1944) estimated that 10,000 pairs nested in 32 colonies on the Maine coast. In response to complaints from fishermen that cormorants were eating too many fish, a control program was initiated in 1944 and continued until 1953. Between the 1940s and 1970s, the population experienced a long period of slow growth, which was largely attributed to the control efforts (shooting, oiling eggs, destruction of young and eggs, and harassment at nesting colonies) and pesticides (Drury 1973, 1974). The cormorant population then underwent a second period of rapid growth from the 1970s through the mid 1980s; in 1977, 15,333 pairs were reported at 103 colonies. In 1984, the population appears to have reached its peak, with 28,760 pairs reported at 121 colonies (Krohn et al. 1995).

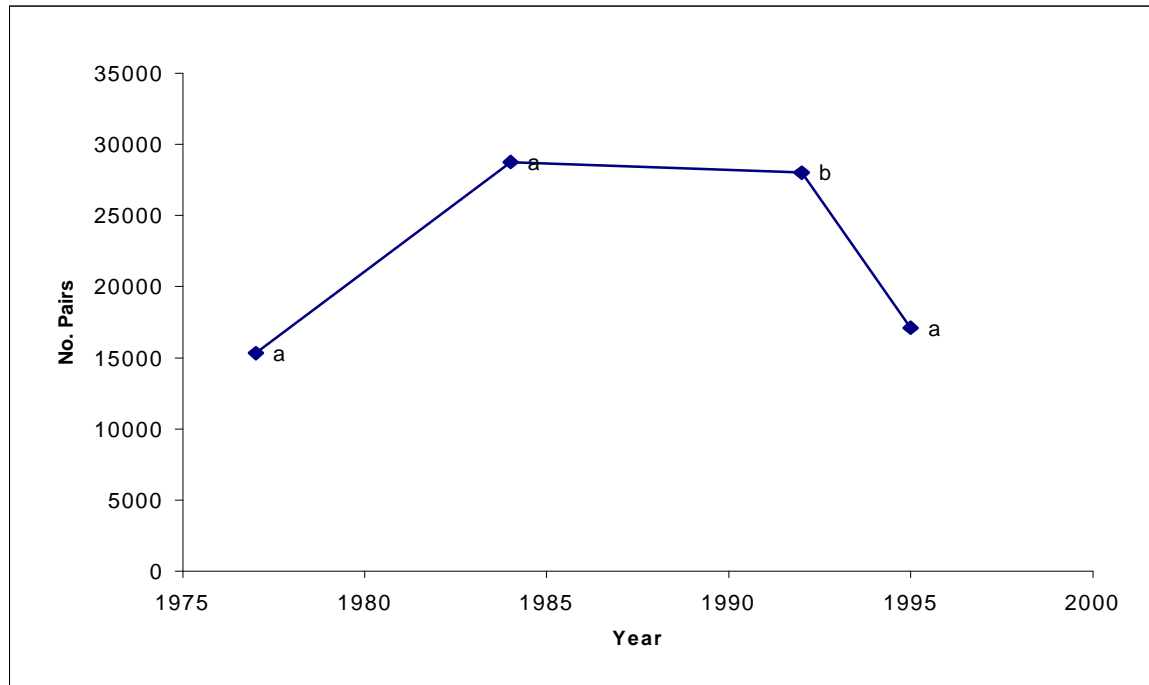
Land Ownership

Maine provided land ownership for all colonies that were over 300 pairs (n = 17); 12 were on private lands owned by individuals (9), or by NGOs such as The Nature Conservancy (1) and The National Audubon Society (2); three were on state lands managed by Maine Dept. of Inland Fish and Wildlife; one was on federal land managed by USFWS; and one was a municipality.

Productivity

No estimates available.

Figure 78. Changes in numbers of breeding pairs in Maine, 1977 – 1994/95.



a = Data from B. Blodget, estimate based on complete surveys.

b = Data from Krohn et al. (1995), estimate based on visits to 67 colonies in early 1990s, about half the number of known colonies.



Maryland

Summary of Population Data and Trends

(Survey completed by D.F. Brinker, Maryland Department of Natural Resources Annapolis, MD.)

Breeding.

The Double-crested Cormorant is a recent breeder in Maryland. First nesting was documented on Poplar Island in Talbot County in 1990; 55 nests were counted (Meritt 1996). Numbers have increased fairly steadily at an average annual rate of 38.7 % since discovery (Figure 79). In 1999, there were four known colony sites, with a total of 1046 pairs. The South Point Spoils colony is the only coastal location; the other three are in Chesapeake Bay (D. Brinker, S. Smith, pers. comm.).

Winter.

No major concentrations are known to overwinter, but small numbers (< 100) do occur. CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified one site, Port Tobacco, where ≥ 100 birds were estimated during at least one count (451 birds in 1998).

Migration.

Thousands of migrants occur in Chesapeake Bay. Peak spring numbers are reported April 5 – 25; peak fall numbers occur October 1 – November 10, and hundreds of thousands are estimated migrating down the coast (D. Forsell, pers. comm.).

Historical Information

Audubon (1843) reported that the Double-crested Cormorant was common as far south as the coast of Maryland in winter. Baird et al. (1884) stated the species was a common spring and fall migrant along the east coast as far south as Maryland. At the turn of the century, the cormorant was reported as an uncommon migrant and winter visitor in the state (Meritt 1996). Later, it was described as an uncommon to common migrant in tidewater areas and in the Allegheny Mountain Section, and rare elsewhere. In recent years it has become more common during the summer in areas of the Chesapeake Bay. Field observers for the Maryland Breeding Bird Atlas Survey predicted the first Maryland breeding record would be obtained during the Atlas Survey years (1983 – 1987); breeding was confirmed in 1990 (Meritt 1996).

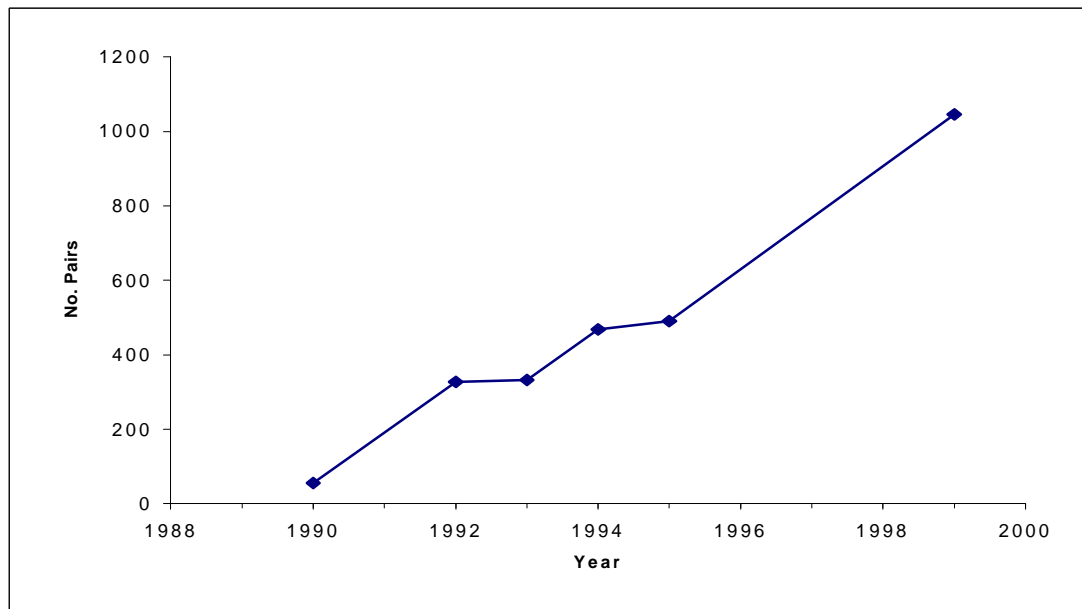
Land Ownership

Three of the four colonies were on state land; Spring Island is on federal land, managed by the Blackwater National Wildlife Refuge.

Productivity

No estimates provided.

Figure 79. Changes in number of breeding pairs in Maryland, 1990 – 1999.





Massachusetts

Summary of Population Data and Trends

(Surveys completed by B.G. Blodget, Massachusetts Division of Fisheries and Wildlife, Westborough MA and L. Henze, USDA/WS, Amherst, MA.)

Breeding.

The Double-crested Cormorant returned as a nesting species to Massachusetts around 1940, possibly earlier (see Historical Information); at that time 53 nests were documented at Shag Rocks in Boston Harbor (Hagar 1941). Since 1977, when 1760 pairs were nesting at 11 known sites, at least three complete surveys have been done. Numbers have increased between 1977 – 1995 at an average annual rate of 8.6 % (Figure 80). At the time of the last survey, 1994 – 1995, 7833 pairs were counted at 27 colonies; there are a total of 28 known sites (B. Blodget, pers. comm.).

Winter.

There were no proven mid-winter records prior to 1976 (Veit and Petersen 1993). With the increase in breeders, birds began occurring in the winter with more frequency, and a small number, probably < 100 – 200 individuals, now winter in the state (B. Blodget, pers. comm.). Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were counted.

Migration.

No significant stopover sites were reported (B. Blodget, pers. comm.), but large numbers have been observed, especially in the fall. In spring, migrants peak in late April, and high counts of up to 1300 have been reported; fall migration occurs September – October, and high counts of 5000 + have been reported (Veit and Petersen 1993).

Historical Information

Bones of adult and young Double-crested Cormorants found in Indian middens indicate the species was breeding on Calf Island, Boston Harbor, as early as 1500 (Hatch 1982). Mendall (1936) notes that in the mid-17th century Roger Williams (1643) stated that “the Indians used quantities of Cormorants for food, returning in the morning, after a night’s hunt, with vast numbers of the birds.” Mendall (1936) thought that Williams’ observations referred to either Massachusetts or Rhode Island, or both. At some point cormorants declined and were entirely extirpated as breeders in New England by the early 19th century. In the late 1800s, the species returned to New England as a nesting species in Maine (Veit and Petersen 1993), but did not nest there with regularity until 1925. It began expanding its range apparently in the 1930s. Hagar (1941) wrote that the colony he discovered in 1940 around Boston Harbor was “probably of several years’ standing;” and noted a communication that cormorants had been breeding at Shag

Rocks since as early as 1937. In the early 1940s, cormorants were regularly summering on the whole of the Massachusetts coast. By the mid 1940s, cormorants were nesting at Shag Rocks, the Weepecket Islands in Buzzards Bay, and on at least one island off Salem (Hatch 1984). Numbers continued to increase in Massachusetts Bay until 1950, when the number of pairs reached 715 (Drury 1973). Numbers in the bay then appeared to decline naturally, which coincided with the control program that was initiated in Maine in 1944 and continued until 1953 (see Maine summary). By 1972, the population in Massachusetts Bay had declined to 325 pairs. Cormorants began a second period of increase, and the Boston Harbor population more than doubled between 1977 and 1981, from 1,000 to 2,510 pairs. By 1982, the entire nesting population of Massachusetts totaled 5,173 pairs (Veit and Petersen 1993). The population has continued to increase, although at a slower rate, and in the latest survey (1994/95) totaled nearly 8000 pairs, as noted above.

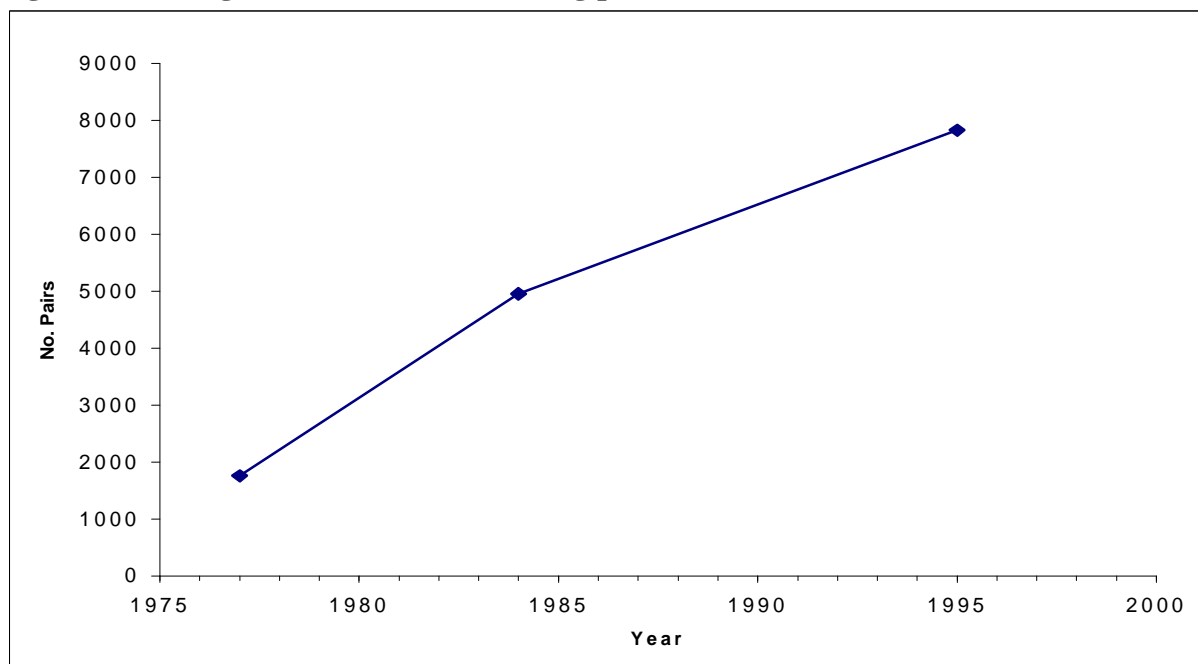
Land Ownership

Ownership information was provided for nine of the 28 breeding sites. Six of the colonies were on public land, while three were on private land. No further information was available.

Productivity

No estimates provided.

Figure 80. Changes in numbers of breeding pairs in Massachusetts, 1977 – 1994/5.



New Brunswick

Summary of Population Data and Trends

(Survey completed by R. Stocck, Maritime Forest Ranger School, Fredericton, New Brunswick; breeding locations from Lock et. al. 1994.)

Breeding.

Korfanty et al. (1997) reported that there are about 15,000 Double-crested Cormorants in the coastal areas of New Brunswick during summer months. In surveys summarized in Lock et al. (1994), 9,278 pairs were estimated between 1986 – 1991 at 15 sites. However, Erskine (1992) reported a minimum of 25 colonies in the province, though information was not provided on when these colonies were active. Overall, numbers are increasing in this province, but the current rate of increase (not available) is less than it was 15 years ago (Korfanty et al. 1997).

Winter.

A small number of Double-crested Cormorants may remain in the province during the winter (Korfanty et al. 1997). Review of CBC data collected during 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were counted.

Migration

No significant stopover sites were reported for migrants, but birds are observed moving along the coastal Bay of Fundy during migration. Peak spring migration occurs mid-late April, with birds observed as early as March 28. Peak fall migration occurs mid-October. A special volunteer project of the Saint John's Naturalist Club has been collecting data on migratory seabirds at the Point Le Preau Bird Observatory for the last five years. The observatory is about 25 miles southwest of Saint John's on the Bay of Fundy. Observations made in spring 1996 – 1998 on heaviest migration days for this species recorded averages of 21 – 56 cormorants per hour. However, many cormorants probably bypass this observation point and migrate overland, so these numbers may not be representative of numbers coming through the entire province. In fall, more cormorants follow the coastline, and averages of 160 – 367 cormorants per hour on heaviest migration days have been observed. However, volunteer coverage has been less intense than in spring; thus more intense coverage may be necessary to obtain more accurate fall estimates (J. Wilson, pers. comm.).

Historical Information

The only early record we obtained for Double-crested Cormorants in New Brunswick was from Herrick (1878), who reported that the species probably nested formerly at Grand Manan, but had ceased to do so. Lewis (1929) reports a colony present at Saltkill, Ids., Lepreaux, estimated at 200 pairs. No other information is available at this time.

Land Ownership

No information available.

Productivity

No estimates available.



Newfoundland

Summary of Population Data and Trends

(No survey was completed for Newfoundland. Distribution and population information were provided by W. Montevecchi, Memorial University of Newfoundland, St. Johns, NF, Korfanty et al. 1997 and Lock et al. 1994.)

Breeding.

In a survey of coastal Newfoundland and offshore islands in 1996, W. Montevecchi reported at least 343 nesting Double-crested Cormorants at a minimum of six locations (*in* Korfanty et al. 1997). At many of these sites Double-crested and Great Cormorants occurred together; DCCOs may also have nested at Great Barasway, but at this location numbers of DCCOs and GRCOs were not distinguished (Korfanty et al. 1997; W. Montevecchi, pers. comm.). Since 1980, 8 – 9 known breeding colonies have been observed (Lock et al. 1994; Korfanty et al. 1997). Numbers are at low density levels in Newfoundland compared to population levels in other Maritime provinces, and it is not known if these low density population levels are stable (Korfanty et al. 1997).

Winter.

Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were estimated during a count, though small numbers were occasionally observed.

Migration.

No information available.

Historical Information

The earliest mention of cormorants in Newfoundland is in 1594, when Europeans visited a small village on the shore of St. George Bay on the west coast of Newfoundland and found “foules called Cormorants, which they [the Indians] had pluckt and made ready to have dressed” (Hakluyt 1904). Lewis (1929) noted that while the specific identity of these birds is uncertain, DCCOs nested here at the time he was writing. Cormorant bones were also abundant at a 17th century Indian site on this province (Montevecchi and Tuck 1987). Audubon (1843) reported a large colony on Cormorant Island as he sailed towards the Straits of Belle Isle. Other early records include a colony at Hawk’s Bay (Reeks 1869). Audubon (1843) also documented human persecution at this time, noting that while “the fishermen and eggers never gather their eggs...they commit great havoc among the young, which they salt for food or bait.” By the time Lewis was writing, Double-crested Cormorants were recorded at eight colonies with < 50 pairs estimated at most of these; for a few of these colonies, no population estimate was available.

Land Ownership

No information available.

Productivity

No estimates available.



New Hampshire

Summary of Population Data and Trends

(Survey completed by J. Kanter, New Hampshire Fish and Game Dept, Concord NH.)

Breeding.

The Double-crested Cormorant has nested in New Hampshire since the 1940s – 1950s. The trend in breeding numbers in this state is not clear. There are only three known colony sites, all coastal. Small numbers of breeding pairs (8 – 24) have been reported since the mid-late 1970s, with small increases in the 1980s to fairly large increases in the early-to-mid 1990s. However, these large increases are thought to be from birds dispersing from Maine to New Hampshire. Numbers declined again in 1998, after the large newly formed colony in New Hampshire at White Island abandoned this site due to harassment employed to preserve the site for breeding terns. In 1998, only 20 pairs were reported at one site (J. Kanter, pers. comm.).

Winter.

No substantial numbers winter in the state (J. Kanter, pers comm.).

Migration.

Peak migration numbers are observed in April and October, but no estimates of numbers available (J. Kanter, pers. comm.).

Historical Information

No information available.

Land Ownership

Ownership information was provided for the three known breeding sites. Square Rock and Lunging Rock, which are next to each other, are both privately owned lands, owned by individuals. White Island, where cormorants were harassed and eventually abandoned the site, is state owned land (J. Kanter, pers. comm.).

Productivity

No estimates provided.



New Jersey

Summary of Population Data and Trends

(Survey completed by D. Jenkins, New Jersey Division of Fish, Game and Wildlife, Trenton,; breeding data provided by J. Walsh, Cape May Bird Observatory, and R. Kane, New Jersey Audubon Society.)

Breeding.

In New Jersey, summering patterns of the Double-crested Cormorant throughout the 1970s and 1980s indicate an increasing population, but breeding was not documented until 1987, when colonies were found in lower Newark Bay. Between 1993 – 1997 an atlas survey was conducted and cormorants were confirmed nesting on artificial structures at five locations. At all five locations cormorants nested on manmade structures. Information on numbers of breeding pairs is not available, but colonies are believed to be small, < 100 pairs (J. Walsh, pers.comm; R. Kane, pers. comm).

Winter.

The DCCO occurs in winter, but no estimates of numbers were available. Leck (1984) reported “small numbers remain into winter.” CBCs conducted between 1959 – 1988 suggested an increase of 3.5 % per year. More recent counts (1984 – 1998) suggest that numbers have continued to increase (Figure 81), though these data have not yet been analyzed for percent change. High statewide counts of > 600 birds were recorded in 1992 and 1998 (Sauer et al. 1996).

Migration.

In the fall, peak numbers of migrants are seen in early-mid Oct, both along the coast and inland. Maximum number recorded was 21,856 on October 17, 1997. Hundreds of thousands pass through during migration, with an average number of 189,768; a maximum of 240,676 was reported by Avalon Sea Watch in 1996. The spring migration peaks in late April or early May, with a maximum count of 3500 birds, Avalon, April 17,1981. Flight is mainly over land, with many birds crossing from Delaware and dispersing northward up the Cape May peninsula (Walsh et al. 1999).

Historical Information

Little historic information on the Double-crested Cormorant in New Jersey was obtained. Stone (1937) summarized records of Double-crested Cormorants observed around Cape May between 1890 – 1934. Most observations were of spring and fall migrants. Sightings were also reported during the summer, but it was not possible to determine if these birds were early migrants or non-breeding birds that had summered in the area. Sibley (1993) notes that the Double-crested Cormorant is apparently more numerous around Cape May than formerly, at

least during the summer, as Stone (1937) only listed five summer records of the species. Most of the flocks reported by Stone (1937) were small, < 100 birds, though “an immense flock stretching well across the sky” was observed in 1929. Numbers of migrants appear to have remained similar since the early 1980s, based on high counts. Leck (1984) reported a high count of 14,000 birds at Cape May on Oct 11 – 12, 1981, and Sibley (1993) reported a high count of 20,000 at Cape May Point on Oct 11, 1983.

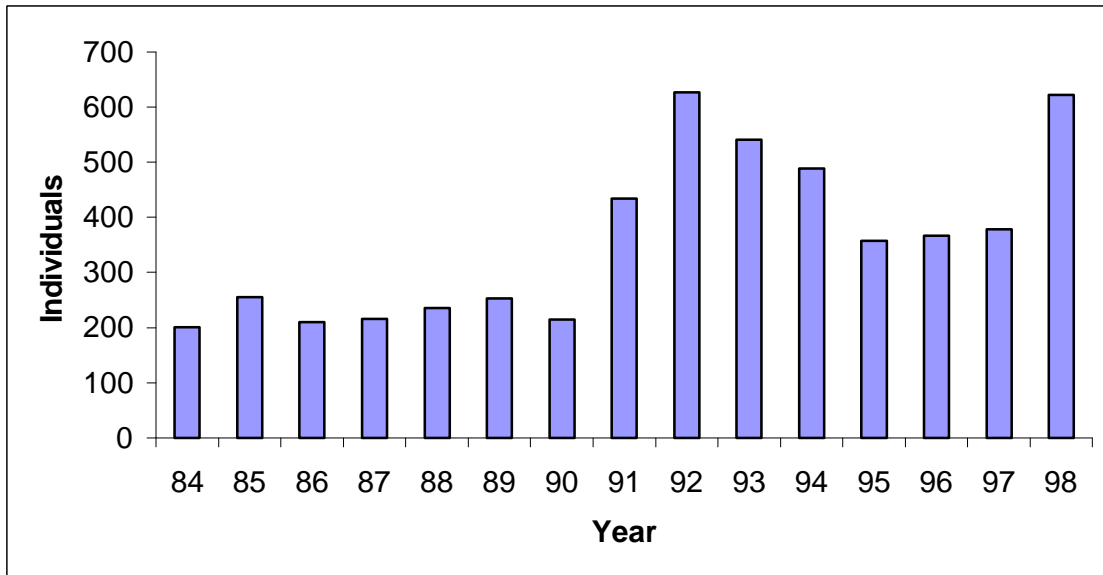
Land Ownership

No information provided.

Productivity

No estimates provided.

Figure 81. December estimates of DCCOs in New Jersey based on CBCs, 1984 – 1998.



New York (Coastal)

Summary of Population Data and Trends

(Survey completed by R. Miller, New York State Dept. of Env. Cons., Delmar, NY.)

Breeding.

The Double-crested Cormorant was first reported breeding in New York in 1945, but was not reported as a breeder in coastal New York until 1977 (see Historical Information, below). Over the last 15 years, numbers have increased significantly in the Long Island / New York City

region. In 1985, there were an estimated 585 breeding pairs in this region; in 1998, there were an estimated 3,274 pairs (R. Miller, pers. comm.), an average annual rate of increase of 14.2 % (Figure 82). In the latter year all known breeding sites were surveyed, with the exception of Plum Island and North and South Brother Islands. North and South Brother Islands were surveyed in 1999, and these counts were figured into the 1998 total (Miller 1998). Excluding Plum Island, nine breeding sites were reported during the last survey, but we only received location information for five. Based on 1997 estimates for inland New York colonies and 1998 estimates for coastal ones, the entire state population (upstate and Atlantic Coast combined) was estimated at 12,346 pairs.

Winter.

Up to several thousand birds winter along Long Island and in the New York City region (R. Miller, pers. comm.). However, review of CBC data collected between 1984 – 1998 reported counts of > 100 birds at only two sites, Queens (103 birds in 1991) and Staten Island (150 birds in 1998) (Sauer et al. 1996).

Migration.

During spring and fall migration, large flocks are observed along the shores of Long Island. On heavy flight days, daily counts will sometimes top 10,000 in both seasons, especially in late September, when fall migration peaks (Miller 1998). Spring migration peaks late March – early April (R. Miller, pers. comm.).

Historical Information

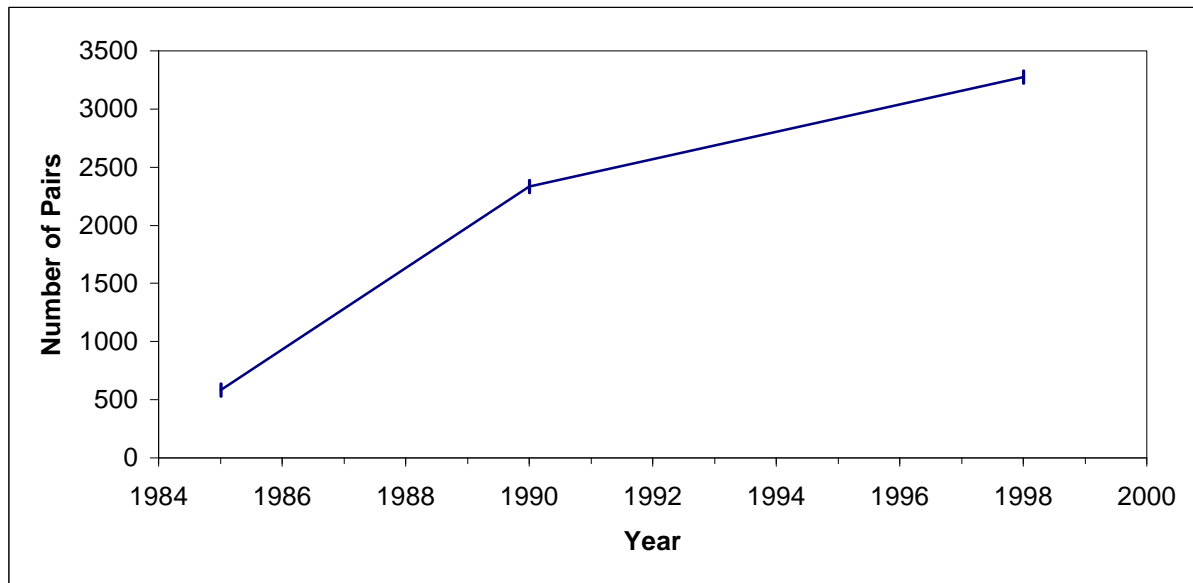
Though the first authentic record of cormorants nesting in New York was not obtained until 1945, and was at an inland location [see summary for New York (Inland)], Arbib (1988) notes that in pre-colonial times the cormorant could have nested within the boundaries of the future New York state, given its distribution along the East Coast. The first nesting for Atlantic New York was not documented until 1977 at Fishers Island (Bull 1981). As increases occurred in the Atlantic Coast population, cormorants moved southward from breeding sites in the Maritimes to Maine, Massachusetts, and eventually to Long Island, NY. In Long Island / New York City region, numbers increased from 585 pairs in 1985 to 3528 pairs in 1995. See summary of New York (Inland) for additional information on DCCOs in New York.

Land Ownership

Ownership information was provided for five of the New York-Atlantic breeding colonies. Three were on privately owned lands, owned by private corporations, organizations and individuals. Two were on public lands that were not identified further (R. Miller, pers. comm.).

Productivity

No information available.

Figure 82. Changes in breeding numbers in New York (coastal) 1985 – 1998.

Nova Scotia

Summary of Population Data and Trends

(Survey completed by R. Stocck, Maritime Forest Ranger School, Fredericton, New Brunswick.)

Breeding.

Population information here is summarized from various sources. The last survey was conducted in 1992; at that time 12,000 pairs were estimated, and the population appeared to be declining (Milton et al. 1995) (Figure 83). A survey was planned for 1997 (Korfanty et al. 1997), but we were not able to obtain information as to whether or not this occurred. A review of surveys at selected colonies suggests that numbers are no longer growing and may be undergoing a small decline (Milton et al. 1995), or may be stabilizing (Korfanty et al. 1997). Lock et al. (1994) reported 69 active colonies observed in 1987 (a complete survey year), and a total of 71 colonies observed between 1987 – 1992. However, Erskine (1992) reported that “recent” surveys by provincial and federal officials indicated at least 80 colonies (the Atlas survey work occurred between 1986 – 1990). The average annual rate of increase between 1972 – 1992 was 5.4 %; however, most growth occurred between 1972 – 1980, when numbers increased at an average annual rate of 14.3 %.

Winter.

Review of CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were estimated during a count.

Migration.

No information available.

Historical Information

There are several early records of cormorants in Nova Scotia from the 17th century. The earliest is from 1604, when Europeans visited the Isle of Cormorants, an island west of Cape Sable on the southwest coast of Nova Scotia, “so named because of the infinite number of these birds, of whose eggs we took a barrel full,” (Champlain 1922). In a footnote to this account, it is suggested that this island was probably the “present [1922] Green Island.” Lewis (1929) notes that in this same account, cormorants are mentioned among the birds found on the present Mud Island group, south of the coast of Yarmouth County. In 1610, cormorants were observed and killed in the same region, off southwestern Nova Scotia, and Lewis (1929) notes that in both accounts, it is a fair probability that at least some of these cormorants were Double-crested. In 1634 – 1635, the Indians of Cape Breton were observed to eat cormorants in summer (Lewis 1929). In 1835, Audubon (1835) reported that “a good number [of DCCOs] breed on Seal Island [near Cape Sable] off the Bay of Fundy.” The next record of cormorants appears to be from the early twentieth century; Lewis (1929) reports a letter dated 1928 which tells of a cormorant rookery that existed “some years ago...on the Bay of Fundy coast, west of the [Digby] ‘Gut’,” Digby County. It is not clear which species of cormorant is being referred to.

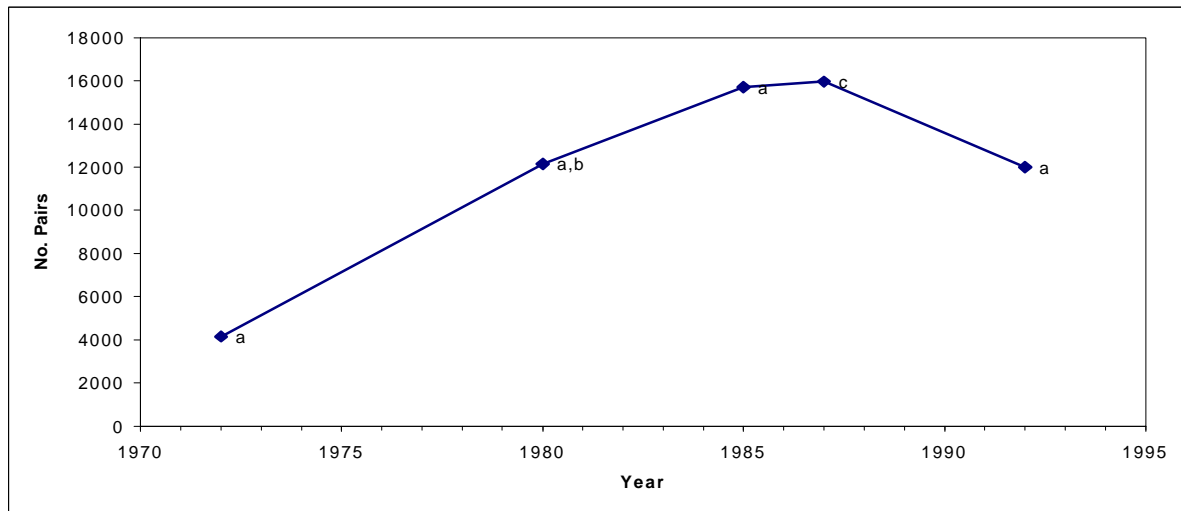
By the early 1900s, or perhaps earlier, cormorants had greatly declined in this province; Korfanty et al. (1997) report that “Double-crested Cormorants were pretty much extirpated from Nova Scotia during the early 1900s.” Lewis (1929) reported that ornithologists visiting islands off the southwest coast of Nova Scotia during the twentieth century found no nesting cormorants; breeding cormorants were reported at only two northeastern locations in the mid-1920s, Bird Islands and Antigonish, with a total of 67 pairs. Numbers remained very low or absent for some unknown amount of time. However, numbers are believed to have increased to 1,300 – 1,500 pairs by the mid to late 1950s, and growth during this period is believed to resemble that of cormorants along the Maine coast. The first thorough survey of Nova Scotia’s colonies was conducted in 1971. During the 10 year period between 1971 and 1980 – 1982, the population nearly tripled, growing from 4,150 to more than 12,000 pairs. In 1985 numbers peaked at 15,700 pairs. Numbers then began to decline.

Land Ownership

No information available.

Productivity

No estimates available.

Figure 83. Changes in breeding numbers in Nova Scotia, 1972 – 1992.

a = Korfanty et al. (1997)

b = Estimate from counts 1980-1982

c = Lock et al. (1994)



Pennsylvania

Summary of Population Data and Trends

(Survey completed by D. Brauning, Pennsylvania Game Commission, Montgomery PA.)

Breeding.

The Double-crested Cormorant appears to be a recent breeder in Pennsylvania; the first nesting was not documented until 1996, when one nest with three young was observed at Wade Island (McConaughy 1996). Since that time, small numbers of birds (≤ 5 pairs) have continued nesting at this site through 1999 (M. Ross, pers. comm.).

Winter.

No significant numbers are reported to winter in this state. CBC data collected between 1984 – 1998 (Sauer et al. 1996) identified one site, Lock Haven-Jersey Shore, where ≥ 100 birds were estimated during a count (194 birds estimated in 1989, but no birds recorded at this site in other years during this period).

Migrants.

Peak spring migrant numbers are recorded in mid-to-late April; fall migration peaks in the first half of October. During these times, hundreds to thousands can be seen passing through the state. Birds do not remain roosting in large numbers on a regular basis, but four areas were

identified where migrants concentrate to feed or roost: the Lower Del River in Philadelphia; the lower Susquehanna River in the Harrisburg area; at Hammond and Cowanesque Lakes in Tioga County, which borders New York; and on Lake Erie, Presque Island State Park. (Ross 1995; D. Brauning, pers. comm.).

Historical Information

Little information available at this time. Mendall (1936) reports that Wood (1935) observed 19 cormorants at Harrisburg on May 3, 1935, which constituted the first record of more than a single individual in this region. Brauning (1992) reported the cormorant as a species observed during the breeding season, but without sufficient evidence to be considered nesting, and remarked that it was likely a candidate for the state's next "confirmed" breeder. The first breeding record was obtained at Wade Island in the Susquehanna River near Harrisburg in 1996 (McConaughy 1996); non-breeding roosting occurred at this same locale a number of years prior to nesting (R. Ross, pers. comm.).

Land Ownership

Land ownership was identified only for three sites: Wade Island is owned by the Pennsylvania Bureau of Forestry; Hammond and Cowanesque Lakes are federal dam projects of the U.S. Army Corps of Engineers; and Presque Island State Park is under state ownership (D. Brauning, pers. comm.).

Productivity

In 1996, three chicks were observed on July 12 in the single nest at Wade Island (McConaughy 1996).



Prince Edward Island

Summary of Population Data and Trends

(Survey completed by R. Dibblee, Dept. of Technology & Environment, Fish & Wildlife Division, Charlottetown, PEI.)

Breeding.

The Double-crested Cormorant has been breeding on Prince Edward Island since at least the 1940s. Yearly surveys for cormorants have been conducted since the 1970s, and all known colonies are counted. At this time, there are 12 known colony sites, with eight of them active. Data were available for 18 surveys since 1976. Since that time numbers have increased at an average annual rate of 13.8 %, though growth has not been steady. Numbers reached their peak of 9,769 pairs in 1997, and then declined in 1998 and 1999 (Figure 84). Korfanty et al. (1997) suggested that growth may be beginning to level off; an average annual rate of increase of 2.8 % over the last decade (1989 – 1999) supports this conclusion.

Winter.

Other than the occasional rare individual, cormorants do not winter in the province (R. Dibblee, pers. comm.). Review of CBCs conducted 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were observed in one year.

Migration.

Large numbers (1000s) of migrants have been reported in Malpeque Bay (North Shore – West), but it is difficult to distinguish between migrants and locally produced birds that are flocking together (R. Dibblee, pers. comm.).

Historical Information

First nesting record was obtained in 1941, at which time there were a total of 75 nests at one known colony site. No additional information is available about possible earlier occurrences in this province (Hatch and Weseloh 1999). Breeding numbers increased very slowly from the 1940s until the late 1970s, when 423 pairs were counted at two of three known colony sites. In the early 1980s numbers began to increase rapidly and new colonies began forming. The most rapid growth occurred between 1983 – 1989, when numbers increased by an average annual rate of 25 %.

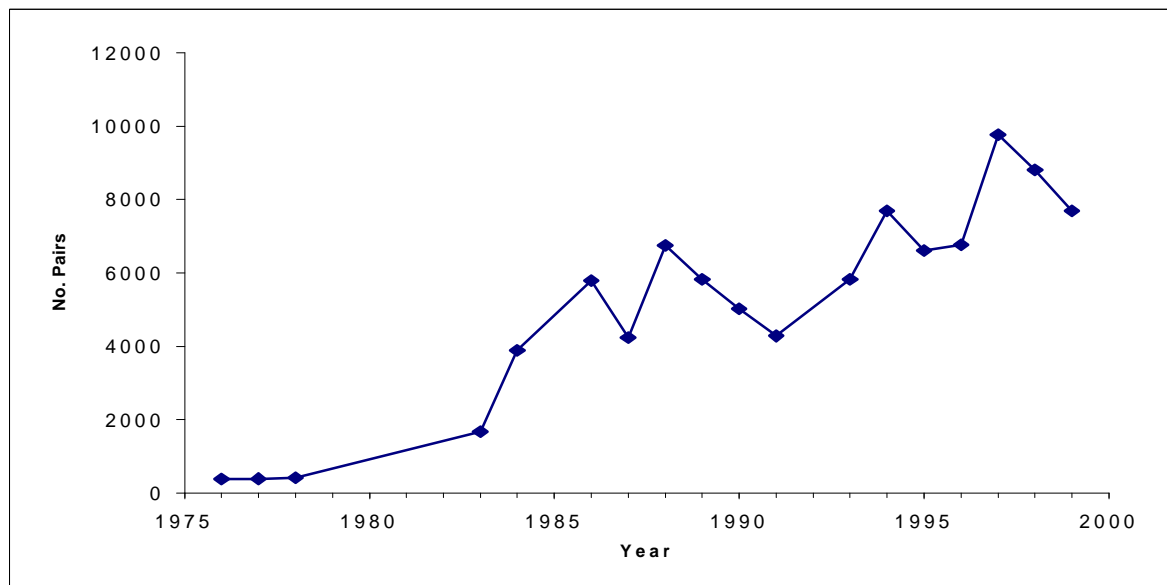
Land Ownership

Land ownership information was provided for six of the breeding colonies. Two were on lands owned by the government of P.E.I.; three were on lands owned by private individuals; one was on land owned by a private organization, the Island Nature Trust. Additionally, one colony was on land that was noted as probably owned by the government of P.E.I. (R. Dibblee, pers. comm.).

Productivity

No information available.

Figure 84. Changes in numbers of breeding pairs on Prince Edward Island, 1976 – 1999.





Québec

Summary of Population Data and Trends

(Survey completed by J.F. Rail, Canadian Wildlife Service, Ste-Foy, QB.)

Breeding.

The Double-crested Cormorant has been breeding on the St. Lawrence River, Estuary and Gulf for hundreds of years (see Historical Information, below). Between 1970 – 1998 a total of 99 colonies were reported on the St. Lawrence River, the St. Lawrence Estuary, the North Shore, the Gaspé Peninsula, Anticosti Island, and the Magdalen Islands. Province-wide surveys estimating colony size in the same year have not been conducted, so it is difficult to establish size and trends for the entire breeding population in Québec. Data were supplied for colonies collected in years around 1970, 1980 and 1990 (e.g., data collected 1967 – 1973 were assumed to represent 1970; 1977 – 1982 represented 1980; 1987 – 1992 represented 1990), and data for all colonies around each year were totaled. Numbers appear to have increased substantially: in 1970, numbers were roughly estimated at 5000 pairs, in 1990, numbers were estimated at about 25,000 pairs (Figure 85). Obviously, these estimates must be interpreted cautiously because cormorants move frequently and size of individual colonies may change dramatically year-to-year (J.F. Rail, pers. comm.).

Chapdelaine and Bédard (1995) provided an estimate for 1990 that was calculated in a similar manner: data collected at colonies (primarily 1986 – 1990) were combined (a few colonies were counted in 1983 and included in this estimate). For the 1990 total, they estimated 27,320 pairs at 70 active colonies. This number was compared to one calculated for the same area in the early 1980s (DesGranges et al. 1984), which estimated 43 active colonies and about 12,000 breeding pairs. Based on these figures, Chapdelaine and Bédard (1995) determined that cormorant numbers increased everywhere in the decade between the late 1970s and late 1980s. The composite annual rates of increase were about 11 % for the Estuary, 16.5 % for the North Shore, 7 % for the Gaspé Peninsula, and 2 % for the Magdalen Islands. DesGranges et al. (1984) concluded that populations had especially increased in the St. Lawrence Estuary between 1965 and 1979, but that the number of colonies and pairs on the North Shore had declined by more than half between 1928 – 1980. Increases may be related to reduction in human persecution and illegal population control, and to a possible increase in the stock of small fish species such as sand lance and capelin owing to commercial overfishing of large predator fish such as cod (Chapdelaine and Bédard 1995). In the last decade population trends in the St. Lawrence River have been reversed due to introduction of a 5-year culling program in 1989. In 1991, the estuary was estimated at 12,081 pairs, a sizable drop since the culling program began (Bédard et al. 1995a). The goal for this area was to maintain the population at 10,000 pairs. However, the program was halted in 1993 because the population fell below the 10,000 pairs threshold, with 9,561 pairs estimated in this year (Bédard et al. 1999).

Winter.

Cormorants are not known to winter in Québec (J.F. Rail, pers. comm.). Review of CBCs conducted 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were observed in one year.

Migration.

The Canadian Wildlife Service did not have any information on major stopover sites for migrating cormorants, nor did they know if such sites existed.

Historical Information

The earliest mention of cormorants in Québec, recorded in 1591 at the Magdalen Islands, is perhaps the earliest reported observation in the range of the Double-crested Cormorant (Lewis 1929). Lewis (1929) notes that “the anonymous author... speaks of a small island called ‘The Isle of Cormorants’ (Hakluyt 1904),” which is probably “the one now known as Shag Island, on the eastern side of the main group of the Magdalens.” Probably both names were used because it was the nesting place of one or more species of cormorant. Other early records (Lewis 1929) include nesting colonies of Double-crested Cormorants along the north shore of the Gulf of St. Lawrence near the Harbour of Great Mecatina Island observed in 1833 (Audubon 1835; 1843); near Sloop Harbor [now (1929) Whale Head] (Coues 1861; Packard 1891); Harrington Harbor (local accounts); Bird Bay, near the eastern end of Anticosti Island (Schmitt 1902, 1904); Bonaventure Island, Gaspé County (Goss 1889); and again Shag Island in the Magdalen Islands (Bishop 1889). However, by the time Lewis was writing, DCCOs had disappeared from all of these locations, mainly due to human persecution and exploitation. Double-crested Cormorants continued nesting at certain other locations in the area, such as those included in bird sanctuary areas set aside for the benefit of other species. Lewis (1929) reported 26 active colonies in the province between 1915 – 1928, with roughly 3300 pairs. Lewis (1929) noted that in the Gulf of St. Lawrence region, “cormorants outside protected sanctuaries supplied fresh meat for Indians, fresh eggs for local residents, and a considerable quantity of food for sled-dogs and captive foxes...and that some colonies lose practically all their young every year.”

The history of the Double-crested Cormorant in Québec during the last century was reviewed by Chapdelaine and Bédard (1995) and DesGranges and Reed (1981), and is summarized here. At the beginning of the century, commercial fisheries regarded the cormorant as a competitor, which triggered studies on its behavior and feeding habits, and cormorant colonies in the St. Lawrence River, Estuary and Gulf were counted and studied. These studies showed that it ate fish of little or no commercial value. Nevertheless, the cormorant was targeted by fishermen, especially during the first half of the century, and certain colonies were prevented from expanding. Legal control was also undertaken in the second half of the century. Island landowners felt that cormorants were damaging natural vegetation, and the Canadian Wildlife Service collaborated with landowners and Québec provincial authorities to control cormorants on Ile aux Pommes in 1954 – 1955, and in 1978 – 1980, but without substantial reduction in numbers.

In the second half of the century, illegal cormorant control became less common, and cormorant numbers increased. As noted earlier, large scale legal cormorant control was again initiated in the last decade in the St. Lawrence Estuary, and has substantially reduced numbers.

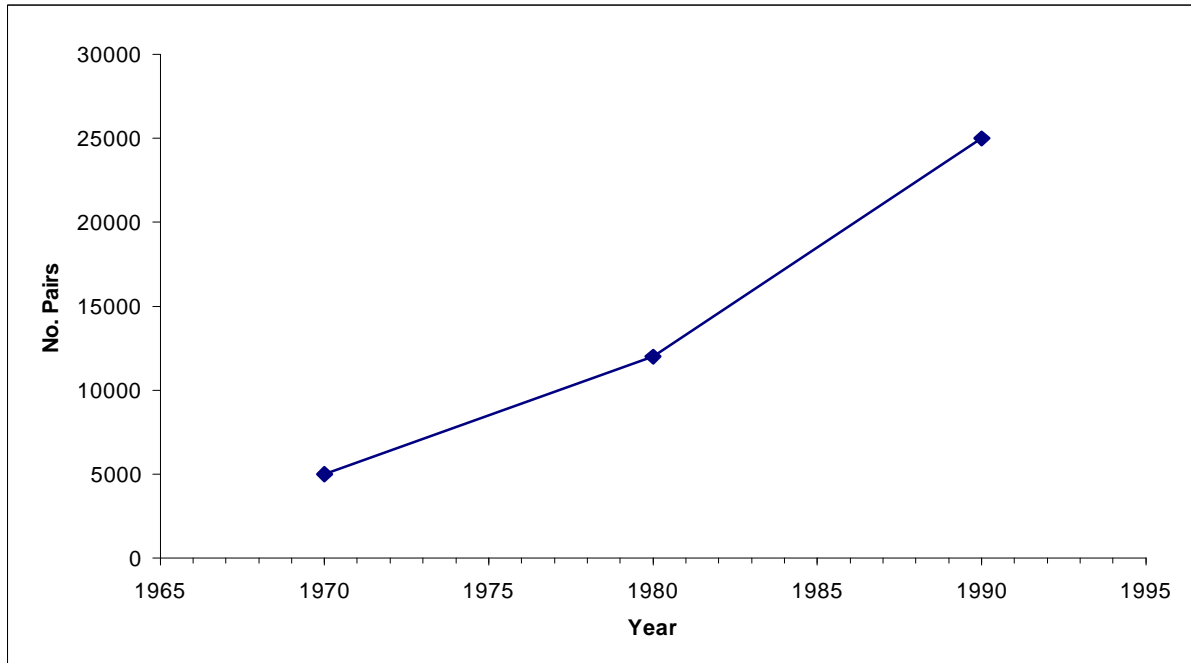
Land Ownership

While land ownership information was not provided for all 122 colony sites, sites that are on protected areas were identified. A total of 54 colonies was located on lands in National Parks, Migratory Bird Sanctuaries, or National Wildlife Areas (J.F. Rail, pers. comm.).

Productivity

Productivity rates ranged from 1.5 – 2.35 fledglings per nest in the Estuary (Cleary 1977; Lemire 1985; both cited in Chapdelaine and Bédard 1995); a rate of 2.87 fledglings per nest was estimated for one colony on the North Shore (Grenier 1987, cited in Chapdelaine and Bédard 1995), while at another North Shore colony (located in Refuge des îles Ste. Marie) productivity was estimated at 2.68 and 2.18 young (13 – 14 days old) per nest in 1996 and 1997, respectively (Rail and Chapdelaine, unpub. data). See Hatch and Weseloh (1999) for additional information.

Figure 85. Changes in breeding numbers in Québec, 1970 – 1990.



Rhode Island

Summary of Population Data and Trends

(Survey completed by C. Raithel, Division of Fish, Wildlife and Estuarine Resources, West Kingston, RI.)

Breeding.

The first known nesting of the Double-crested Cormorant in Rhode Island was documented in 1981 on the Sakonnet Islands (Ferren and Myers 1998a). In 1997, there was a total of 2058 pairs counted at nine colonies. In 1998, a new breeding site was discovered, bringing the total number of known colonies to 10. All colonies are located on islands above Rhode Island Sound and on the Sakonnet River. Additionally, 1 – 2 pairs have been observed breeding on Seekonk River (North) pilings and (South) platform. However, we did not include these latter sites in the total number of breeding colonies listed in Table 6 because they were single or double pair sites and may not be used regularly (C. Raithel, pers. comm.). Since initial discovery in 1981, numbers increased steadily through 1995, then showed small declines in 1996 and 1997 (Figure 86). Much of the most rapid growth occurred between 1981 – 1990, when the average annual rate of increase was 63 %. Between 1991 – 1997, the average annual rate of increase slowed substantially to 4.3 %. Rapid growth has occurred at the West Island and Little Gould Island colonies (C. Raithel, pers. comm.).

Winter.

A small number of birds, about 200, winter in the state (C. Raithel, pers. comm.). Favored overwintering spots are Galilee Harbor and lower Salt Pond (Ferren and Myers 1998a). Review of CBCs conducted 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were observed in one year.

Migration.

Spring migration occurs mainly in April, and peaks between April 20 – 30; fall migration occurs the last week of September – first week of November, peaking about mid October, when as many as 20,000 birds a day are counted (Ferren and Myers 1998a). Up to 100,000 birds are estimated to pass through, feeding and roosting mainly along the coast.

Historical Information

Ferren and Myers (1998a) reviewed the historical distribution of the Double-crested Cormorant in Rhode Island and their work is summarized here. Prior to the arrival of Europeans, the Double-crested Cormorant could have been present in what is today Rhode Island; in pre-Columbian times the food supply was richer and more dependable than it is today, and nesting habitat was probably similar in terms of vegetation and freedom from mammalian predators. However, Native Americans could easily reach likely nesting locations in pre-colonial Rhode Island, as they did in other parts of New England, and any sizable cormorant colony would not have been ignored (see Massachusetts profile, Historical Information). It is fairly certain that cormorants were not present in the state in any significant numbers after European settlement, and the ensuing period of human exploitation of waterbirds all along the Atlantic seaboard. Ferren and Myers (1998a) note that Sturtevant saw only a single cormorant during all his pre-1900 summers in the Newport area. Additionally, Hathaway saw only a single bird in August 1903. Gradually, however, in the first third of the twentieth century, sightings began to increase

and small numbers of summering individuals and immatures were observed on coastal rocks and pilings. Numbers grew slowly; in the 1950s counts rarely exceeded 50, but by 1971, 170 non-breeders were counted on the Sakonnet islands. By 1980, the state's non-breeding summering population was estimated to have reached a minimum of 1,000 birds. The following year, breeding was documented at Sakonnet, where numbers increased rapidly and breeding pairs soon spread to other islands.

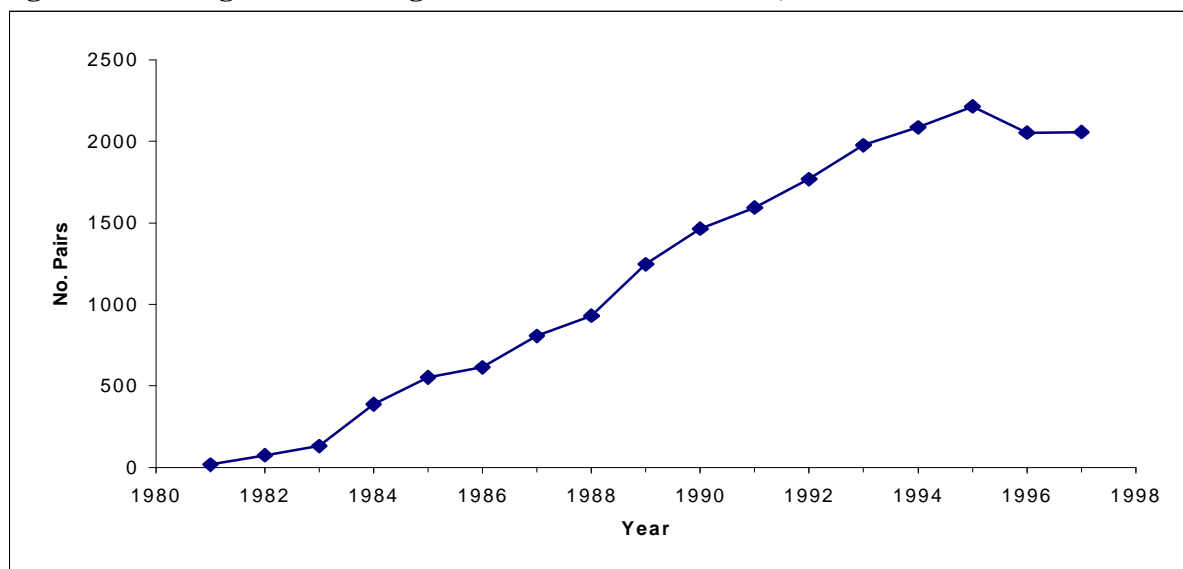
Land Ownership

Of the 12 known breeding sites, 10 were on privately owned lands; Little Gould Island is owned by the Audubon Society of Rhode Island. The other two colonies, Big Gould and Hope Island, are on state land (C. Raithel, pers. comm.).

Productivity

No estimates available.

Figure 86. Changes in breeding numbers in Rhode Island, 1981 – 1998.



Virginia

Summary of Population Data and Trends

(Survey completed by D. Schwab, Virginia Dept of Game and Inland Fisheries, Williamsburg, VA.)

Breeding.

The Double-crested Cormorant may be a fairly recent breeder in Virginia, with first known nesting documented in the James River 3 km east of Hopewell in 1978 (Blem et al. 1980). However, breeding may have occurred historically (see Historical Information). In 1997, there was only one known colony on Shank / Cheasman Island in Chesapeake Bay, with an estimated 60 – 70 pairs. This colony appears to be slowly increasing; in 1993, the only other year for which data were reported, 12 nests were counted (D. Schwab, pers. comm.).

Winter.

In winter, birds can be observed along the James River in the southern end of Chesapeake Bay (D. Forsell, pers. comm), but no estimates of wintering numbers were provided. CBCs conducted between 1959 – 1988 suggested an increase of 12.6 % / year; more recent counts (1989 – 1998) (Sauer et al. 1996) suggest that numbers have continued to increase (Figure 87), though these data have not yet been analyzed for percent change.

Migration.

During migration, very large numbers reported; in the spring, estimates at Fisherman Island Refuge on the eastern shore of Chesapeake Bay range from 20,000 – 30,000 to as many as 300,000 (D. Forsell, pers. comm.). Large numbers also observed in fall, September – October, with birds roosting and feeding in coastal waters and in the marshes and rivers around Chesapeake Bay (D. Schwab, pers. comm.).

Historical Information

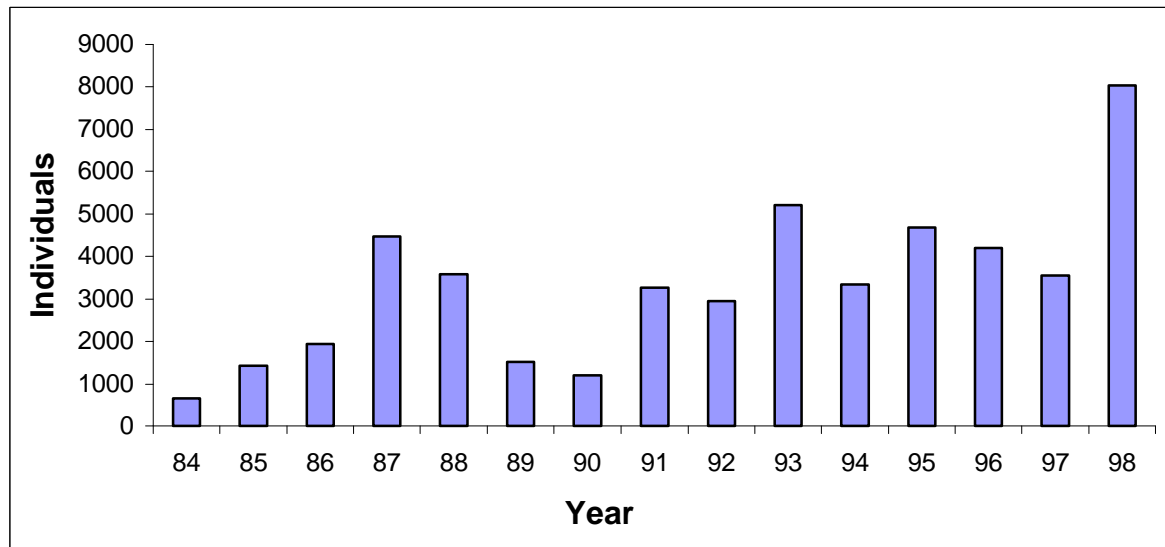
Lewis (1929) noted several 17th century references that mention the abundance of cormorants in Virginia; one reports cormorants among the birds in Virginia rivers “in such abundance as are not in all the world to be equaled.” Lewis (1929) reported that probably most or all of these cormorants were Double-crested. The next records of the species we were able to obtain were from the 20th century (Mendall 1936). These include a small group (10 birds) of non-breeding individuals observed in the summer of 1921 at Wallops Island; an account of a cormorant found in 1924 in the interior of the state that apparently struck a chimney; and an account of a flock of 175 cormorants observed around 1928 during fall migration flying in a southerly direction offshore of Cobb’s Island. We found no records of the species in the 18th and 19th centuries; the species may have been absent or extirpated from the state during this time, as it was across New England (see Zone 5: Summary of Population Data for the Northeast Atlantic Coast Summary).

Land Ownership

No information provided

Productivity

No estimates available.

Figure 87. Annual totals of DCCOs in Virginia based on CBCs, 1984 – 1998.

West Virginia

Summary of Population Data and Trends

(Survey completed by D. Jones, West Virginia Nongame Wildlife & Natural Heritage Program, Elkins, WV.)

Breeding.

The DCCO is not known to breed in West Virginia (D. Jones, pers. comm.).

Winter.

Smith (1996) reports the species as fairly common in the state, September through May. He notes that the species has increased in recent years, with observations at various locales becoming frequent after 1990. In early January of 1995, a count of nearly 50 was made along the Ohio river from Nitro to Robertsburg, suggesting that the species is widespread in small numbers during the winter months. Review of CBCs conducted 1984 – 1998 (Sauer et al. 1996) identified no sites where ≥ 100 birds were observed in one year.

Migration.

During spring and fall migration, thousands are seen along the Ohio River, with peaks observed late April and early May, and again in September and early October (P. Morrison, pers. comm.). Migrants are also seen along the Potomac River; in spring migration, hundreds are reported (K. Leo, pers. comm.). In summer, non-breeding birds are also sometimes observed along the Ohio river with juveniles, probably 1 year old birds (P. Morrison, pers. comm.).

Historical Information

Hall (1983) reported that the Double-crested Cormorant was a fairly regular migrant in small numbers at Cheat Lake, Monongalia County, until the late 1960s. Spring dates were from mid-April to mid-May; fall dates were from late October to early December. The species was much less common elsewhere in the state, but there are scattered records from other locales. Hall (1983) reported that since the mid-1960s very few reports of cormorants were made, which paralleled the general decline in the population concomitant with pesticides in the aquatic food chain. However, he also noted that declines in reports may be due to the fact that the portion of the lake where most cormorant sightings were observed became inaccessible by shore. In Kanawha Valley, where sightings have increased, there were few records prior to 1990 (Smith 1996).

Land Ownership

No information available.

Productivity

Not applicable.



PART III: DIET

INTRODUCTION

In this section, we address several aspects of DCCO diet, and begin by defining relevant terms. *Diet* refers to the proportions of diverse prey consumed by cormorants but provides no measure of impact. *Daily food intake (DFI)* refers to the amounts of prey consumed over a 24-hour period. *Impact* is the effect, if any, of consumption of fish by birds on the fishery in question. Because there are numerous compensatory effects of prey that may eliminate or reduce potential impacts, impact is the most difficult measurement to quantify.

Common names of fishes are not traditionally capitalized in publications (American Fisheries Society 1991), whereas standard format for bird common names requires use of capital letters (American Ornithologists' Union 1998). Use of these two different formats respects nomenclature decisions followed by these two professional societies.

Diet Studies Reviewed

Trapp et al. (1999) reviewed 25 major studies on the diet of the DCCO conducted in 13 states and provinces, 1923 – 1994. We identified an additional 16 major studies conducted in 16 states and provinces; 12 of these were done between 1995 – 1999 (Neuman et al. 1997; Johnson et al. 1999; Belyea et al. 1999; Bur et al. 1999; Blackwell et al. 1995, 1997; Hill et al. 1997; Milton et al. 1988; Rail and Chapdelaine 1998; Glahn et al. 1995, 1998; Roby et al. 1998), while four were conducted prior to 1995 (Robertson 1974; Ainley et al. 1981; Kehoe 1987; and Bivings et al. 1989). We also summarized two reviews / compilations of diet studies conducted in coastal New England, the Maritime Provinces, and eastern Québec (Cairns 1998), and in Wisconsin on Lakes Superior and Michigan, 1983 – 1997 (Matteson et al. 1999). We organized these studies and reviews by prey species groups in four regions: the Atlantic Region (Maine, New Brunswick, Nova Scotia, Prince Edward Island, and Québec); the Great Lakes (Canadian and U.S.); the Southern U.S. (Texas, Mississippi, Alabama and Arkansas); and the Pacific Coast (British Columbia to Sonora, Mexico) (Table 7 – Table 25). Because earlier diet studies were summarized elsewhere (Hatch and Weseloh 1999; Trapp et al. 1999), and because we are interested in current dietary trends for the purpose of this status assessment, we utilized diet studies conducted primarily over the last decade. However, to represent each region as fully as possible, we also include some of the studies summarized elsewhere (Robertson 1974; Ainley et al. 1981; Craven and Lev 1987; Birt et al. 1987; Ludwig et al. 1989; Campo et al. 1993; Neuman et al. 1997; Blackwell et al. 1995, 1997).

Diet Assessment Methods: Advantages and Shortcomings

Most information on cormorant diet was obtained from pellets or regurgitations collected in nesting colonies (especially fish otoliths within colonies). Stomach contents are reported less frequently. Occasionally tags of stocked fish are recovered. All methods have advantages and disadvantages. During the 4th European Conference on Cormorants (Bolgna, Italy, 1995), a group of 28 cormorant researchers, mostly from Europe, discussed methods of assessing cormorant diet and food intake. They also evaluated associated sources of error, attempts to quantify them, areas of consensus, and gaps in current knowledge. Carss et al. (1997)

summarized the results of these deliberations. We relied heavily on this summary, along with remarks from North American investigators studying DCCO diet, to review the advantages and disadvantages associated with several dietary assessment methods. We provide this review to enhance understanding of conclusions reached in the various studies reviewed.

Pellets

Neuman et al. (1997) state that currently, pellets seem to provide the most cost effective, representative, and non-intrusive means of assessing cormorant diet composition. Additionally, in a comprehensive review of diet methods, Carss et al. (1997) note that pellets provide large samples that can be collected quickly with little or no disturbance to birds, and that analysis is fairly easy.

However, Carss et al. (1997) also point out that “pellet analysis is a useful method of obtaining a rough index of cormorant diet in qualitative terms but there is serious doubt as to whether it can be used to derive quantitative information on, for example, species composition or size-range of fish taken.” Carss and colleagues outline several problems and biases associated with pellet analysis. One of the most important is that the diagnostic hard parts recovered from pellets, namely the sagittal otoliths of teleost fish, are often eroded. Differential loss and digestion of otoliths makes identification of some fish species problematic or impossible. For example, Craven and Lev (1987) were unable to identify ninespine sticklebacks in the pellets of cormorants nesting at Eagle Island, Lake Superior, even though this species was abundant in the waters around this site, and most likely consumed by cormorants at the time of the study; this species was identified in regurgitations.

Studies on both captive and wild cormorants suggest that estimating consumed prey size from otoliths recovered from pellets is highly inaccurate, as otoliths of some fishes are “rapidly attacked” and eroded by bird stomach secretions (Duffy and Laurenson 1983; Jobling and Breiby 1986; Johnstone et al. 1990; Harris and Wanless 1993). Most “lost” or eroded otoliths are from the smallest fishes so recovery is size related (Johnstone et al. 1990; Carss et al. 1997), and can result in underestimation of smaller fish consumed and impacts to fisheries. Additionally, age of prey fishes aged by eroded otoliths is likely underestimated (Harris and Wanless 1993). Therefore, attempts to develop an age distribution of fish consumed by cormorants through an analysis of otoliths recovered in pellets may underestimate both older and younger fish. To minimize this bias and determine the most reliable otolith predictors of fish length, some investigators have related otolith erosion levels and size, and developed an otolith classification scheme according to degree of erosion (Adams et al. 1999; Ross and Remaley, in review). However, some scientists (Johnstone et al. 1990; Harris and Wanless 1993) caution that attempts to sort out eroded from uneroded otoliths may be misleading, because erosion sometimes deeply etches otolith details making eroded otoliths appear fresh and indistinguishable from uneroded ones.

An additional problem associated with otoliths is that of secondary consumption (i.e. otoliths in sample represent diet of cormorant prey). Blackwell and Sinclair (1995) caution that a species well represented by otoliths can occur solely as a result of secondary consumption (in both regurgitations and pellets). Johnson et al. (1997) concluded that consumption of invertebrates by DCCOs might be overestimated in cases where primary and secondary consumption products cannot be separated. Pellets may also overestimate primary invertebrate consumption and underestimate soft-bodied fish (Brugger 1993; Neuman et al. 1997), as hard

parts of animals such as crustaceans remain undigested while soft-bodied fishes can be completely digested relatively quickly.

A common assumption about pellets is that they integrate meals over an entire day, and on average are produced once a day (e.g., Craven and Lev 1987; Orta 1992; Derby and Lovvorn 1997a; Johnson et al. 1999). However, research indicates that pellets may integrate meals over a substantial time period, and a single pellet should not be assumed to represent one day's food intake, as less than or greater than one pellet / day may be produced (Duffy and Laurenson 1983; Johnstone et al. 1990; Brugger 1993). More recently, observational and experimental studies with European Shags (*P. aristotelis*) and Great Cormorants in Europe indicate that the assumption of one pellet / day overestimates the rate of pellet production for these species. In south-east Scotland, Russell et al. (1995) found that pellets were produced by shags during the breeding season much less frequently than one / day; on average, a shag produced a pellet once every four days. In Germany, Huppopp and Frundt (2000) found that Great Cormorants observed at winter roost sites regurgitated on average 0.74 pellets / night (range 0.33 – 1.0), about 25 % < generally assumed. These results have important consequences for the use of pellets to estimate energy requirements of phalacrocoracids, because such models have generally assumed that pellet contents give an estimate of daily food intake (e.g., Linn and Campbell 1992; Johnson et al. 1999). Less frequent pellet production suggests that models assuming a production rate of 1.0 pellets / day may overestimate fish consumption and consequently impacts to fisheries.

To enhance the use of pellets as a dietary assessment method and to reduce the biases associated with pellet analysis, Carss et al. (1997) suggest that feeding trials be undertaken and attempts made to quantify size- (or species-) related differential recovery. Specifically, “such trials should involve feeding captive birds on fishes of known species, length, and mass and should take into account the possible influences of stress, deliberate disturbance (on pellet content), and activity.” Such trials may also produce necessary information on pellet formation and the relationship between degree of digestion and quality of fish.

Regurgitations

Regurgitations (boli), compared to pellets, are cheaper and faster to analyze, and the results are much less influenced by differential digestibility of prey. Regurgitations often provide fresh or relatively undigested specimens, that can be measured for more precise estimates of length and age class of fish consumed by cormorants, and for prey biomass or energy consumed. Prey in freshly regurgitated boli generally can be identified at a glance and easily numbered; it is fairly easy to obtain a good measure of the biomass or volume represented by each taxon (e.g., no large inherent bias). Additionally, very small and / or soft-bodied prey that are poorly represented or absent in pellets are generally found in regurgitations. (J.F. Rail, pers. comm.).

However, regurgitations contain the contents of recent meals only, and thus may not provide information on daily diversity within the diet. For example, Neuman et al. (1997) compared prey items found in boli and pellets that were collected on the same day at the same colony, and found large differences in the percent frequency of food items indicated by both methods (Table 11, Table 13). Additionally, regurgitations do not necessarily provide an accurate representation of adult diet, because generally the largest part of the sample is produced by young cormorants (J.F. Rail, pers. comm.), and adults may eat low quality food themselves and feed higher quality food to chicks (Harris and Wanless 1993). Furthermore, studies that utilize otoliths obtained from regurgitations may encounter biases similar to those encountered in pellet analysis. For example, because adult birds swallow fish head first, the fish heads will be

the most digested parts; some otoliths within them may be eroded or not passed in the regurgitation (Carss et al. 1997). Other disadvantages include a greater amount of time required to collect an equal sample size of regurgitations as pellets, and greater caution is required to minimize disturbance while gathering regurgitations (J.F. Rail, pers. comm.). In addition, obtaining complete regurgitations from tree nesting colonies is difficult because whole fish may fall from the nest or small parts may become tangled in tree branches and leaves (M. Bur, pers. comm.). Overall, however, this method may be very effective during the chick rearing period, providing high quality data with possibly fewer biases and problems than pellet / otolith analysis (J.F. Rail, pers. comm.). Although collecting pellets or regurgitations may not cause direct mortality, it is important to note that entering active colonies flushes adults and leaves eggs and chicks vulnerable to predation which may reduce colony-wide reproductive success.

Stomach Contents

Stomach contents (which may be collected from the esophagus, proventriculus and gizzard) are most commonly obtained from dead birds but are sometimes also acquired by flushing stomachs of live birds. Advantages of studying diet from stomach content analysis include: sample is often fresh; it is easy to distinguish recent prey from partially digested fishes and prey retained in bottom of stomach. Thus some of the biases associated with pellets can be avoided. Stomach contents can also be accompanied by site specific and demographic information (Carss et al. 1997).

However, there are several important disadvantages associated with this method. First, collecting DCCOs for stomach contents usually requires killing birds. While stomach contents can be obtained from live birds by flushing stomachs with stomach pumps and emetics, these procedures are stressful for birds (Duffy and Jackson 1986; Harris and Wanless 1993). Additionally, both methods are time consuming and / or costly (Hatch and Weseloh 1999). Due to these problems, sample sizes are usually small and further reduced if some stomachs are empty (Carss et al. 1997).

DFI cannot be determined from stomach content analyses, because it is not possible to determine if a bird has finished feeding for the day (Carss et al. 1997). Additionally, this method is likely to under sample known spatial and temporal variation in prey captured each day (Hatch and Weseloh 1999). Foraging behavior of birds may also lead to biased assessments of diet. For example, patch foraging may create a non-random distribution of prey among stomachs; birds foraging individually (generally thought to take larger fish) are thought to be more vulnerable to shooting than birds foraging in groups (presumed to take smaller fish) (Carss et al. 1997). However, shooting incoming birds (returning from foraging trips) at random locations from a boat-blind 150 – 200 m distant from the island colony reduces variation and randomizes birds that may be foraging individually or in groups (M. Bur, pers. comm.).

While it may be possible to obtain more precise information about species and sizes of fish eaten with this method (Harris and Wanless 1993), the more digested the stomach contents are the greater the potential for bias, at least theoretically, as some items will be more resistant to digestion than others. Finally, studies estimating diet from intact items alone will underestimate the proportion of small fish species and overestimate the average size for some larger ones (Carss et al. 1997).

To enhance use of this method, Carss et al. (1997) suggest that a sample from one localized area is more informative than one from a whole “catchment”, and that small samples from a restricted time and place are more useful than if spread out over an entire year or

watershed. Carss et al. (1997) also propose a standardized technique for analyzing stomach contents, and include suggestions for potential key bones to be used in analysis of dried remains, which include atlas, thoracic, and caudal vertebra, pharyngeal teeth, pelvic girdles, opercular bones, cleithra, lower jaws, and cyclostome teeth. Key bones must be robust and resistant to digestion, relatively easy to identify, and diagnostic; the only published examples are salmonid atlas and eel thoracic vertebrae. Otoliths were rejected because their presence did not correspond well with the occurrence or size of other bones from the same species.

Recovered Tags From Stocked Fish.

At some colonies, tags recovered from stocked salmon smolts were used to determine predation. One drawback associated with this method is that converting tag recovery data into estimates of diet composition is difficult because tag recovery rates are hard to measure and capture of tagged fish may not be representative of untagged fish of the same species (Cairns 1998). However, Ross and Johnson (1999a) found that previous analyses of lake trout stocking losses benefited from the return of coded wire tags, in addition to diagnostic hard parts, recovered in pellets.

Conclusions: Diet Assessment Methods

When attempting to determine diet composition, the method of diet assessment needs careful consideration, and biases associated with each method should be well understood and compensated for. Depending on the circumstances (e.g., colony size, accessibility, reaction of birds to intrusion, etc.), and goals of the study, one method may be much more appropriate than another, and limitations of the method should be considered in study design (J.F. Rail, pers. comm.) and interpretation of results. Many studies are designed to assess impacts to fisheries. To assess these impacts, solid information is needed not only on cormorant diet, but also on daily energy requirements. Because of associated biases, “good” estimates of daily food intake cannot be obtained through pellet analysis, regurgitations or stomach contents of shot birds. Future estimates of daily food intake should be based on considerations of the energy requirements of wild birds (Carss et al. 1997). Methods to measure these are described in PART VII.

RESEARCH NEEDS.

Measurements Used to Report Diet

In addition to the method used to assess diet, an important factor in all studies is the measurement used to report diet. Measurements, along with assessment methods, vary, and this can affect results, conclusions, and comparability among studies. In the diet studies we reviewed, five measurements were reported: *percent frequency*, defined as the proportion of samples containing a prey taxon; *percent number*, defined as the number of specimens of a taxon as a percent of all specimens in a sample; *percent biomass*, defined as the biomass of a taxon as a percent of total sample biomass; *percent volume*, equivalent to percent biomass, except that quantities are measured volumetrically; and *relative importance*, defined as the sum of the percent of each taxon divided by the number of samples analyzed, was reported in one study. Each measurement reflects different but important aspects of prey utilization.

All of these measurements can be confounded by differential digestibility or variation in size, availability, or abundance of prey taxa (Blackwell et al. 1995). Robertson (1974) suggested that biomass is probably a more critical indicator of energy flow than number of individual food items taken, but this measurement, in isolation, provides no information on the frequency with

which the taxon is taken or what proportion of the diet consists of the taxon. Because compensatory mechanisms may operate more strongly on different age classes, and because number of fish taken does not indicate proportion of each age class available, simply reporting number of fish taken does not indicate impact to a fishery. Studies reporting numbers of fishes consumed need to include estimates of size of fishes taken, because the removal of even large numbers of very small fishes may not significantly affect a fishery. For example, the loss of salmon fry or small parr could result in enhanced survival or growth of remaining fish, while removal of large parr or smolts could result in a reduction in the numbers of juvenile fish reaching the sea and perhaps in the numbers returning as adults (Carss and Marquiss 1997). Similarly, in Oneida Lake, consumption of age 0 yellow perch and walleye is buffered by compensatory effects, and essentially has no effect on the resulting number of fish reaching the fishery, while consumption of age 1 and older fishes of these species is believed to represent additive mortality, making fewer fish available to the fishery (C. Adams and L. Rudstam, pers. comm.). Therefore a combination of measurements is necessary to accurately reflect the importance of and impacts associated with a particular prey species within the diet. Reintjes and King (1953) suggest that a prey taxon ranking high in frequency of occurrence, volume, and number is important at the time and place sampled.

RESULTS OF DIET REVIEW

Regional Diet Descriptions

An important factor to consider in the following descriptions of cormorant diet in each region is that most of the diet studies we reviewed were designed or undertaken to assess the impact of cormorant predation on sport or commercial fish species, or to determine the importance of sport or commercial fish species within the diet (Belyea et al. 1999; Ross and Johnson 1999a,b; Johnson et al. 1999; Blackwell et al. 1995, 1997; Milton et al. 1995; Hill et al. 1997; Bivings et al. 1989; Campo et al. 1993; Glahn et al. 1995, 1998; Roby et al. 1998). Such studies may not generate data representative of DCCO diet in general. For example, studies that utilize data from cormorant stomachs that were collected at smolt stocking sites or catfish farms may find predictably high frequencies of these species in the diet. Furthermore, these data often involve only a few hundred DCCOs while the total population in the particular state or province where the samples are collected may consist of many thousands of birds [many of which may not participate in movements to these feeding grounds (Cairns 1998)]. Therefore, to avoid over-representing the importance of results from preferential sampling, we caution that the results we present to describe the diet of DCCOs in each region should be considered in the context of the conditions under which the study was conducted, and the size and status (breeding, wintering, migrant) of the cormorant population described (J.F. Rail, pers. comm.).

Diet on the Pacific Coast

We incorporated data from three studies that examined diet of DCCOs on the Pacific Coast; only one was conducted in the 1990s. Diet in this region consisted almost exclusively of fish, though small amounts of shrimp were also consumed. To categorize diet, we developed four categories of important prey fish in this region: salmonids, estuarine and or littoral species, littoral-benthic species, and pelagic species (Table 7 – Table 10). Though information on diet in several areas was included, much of the data we obtained comes from two main locations, the Columbia River estuary and Mandarte Island, British Columbia. As on the Atlantic Coast, diet

varied regionally to a great extent, and no one species emerged as the most important. However, some fish species were very important in certain regions and across the region generally. For birds breeding in the Columbia River estuary, salmonids were the most important prey item by percent biomass, particularly during the month of May, and were more important overall in the diet of birds nesting further upriver than birds nesting closer to the river mouth (Table 7). Littoral or estuarine species, such as stickleback and peamouth, were the second and third most important species, respectively, by percent biomass. Littoral-benthic species, such as sculpin, were also important prey for the estuary birds (Table 8, Table 9). Along the coast and in the Gulf of California, Ainley et al. (1981) examined DCCO diet at seven sites, and reported that cormorants ate mostly schooling prey that occurred from the surface to near, but not on, bottoms having no relief. In general, littoral or estuarine fish, such as shiner perch and other embiotocids, were important in the diet of cormorants in many areas of the Pacific Coast (Table 8). Littoral-benthic fish, such as sculpin, gunnel, snake prickleback and sucker, also made up a significant portion of the diet (Table 9). Locally important species included Pacific sand lance for birds nesting in British Columbia (Table 8), and clupeids of the genus *Opisthonema* (Table 10) for birds breeding in Sonora, Mexico.

Diet in the Great Lakes

In the Great Lakes region, the studies we reviewed indicate that diet in this region consists almost exclusively of fish, though crustaceans were also consumed in small proportions. To describe the principal fish species consumed, we developed five freshwater fish categories (B. Vondracek, pers. comm.): open-fresh water fishes, littoral-fresh water fishes, bottom dwelling-fresh water fishes, salmonids, and locally important fishes (Table 11 – Table 15). Fish species from the open-fresh water category, specifically alewife and gizzard shad, appeared to be most important (*importance* here is defined in terms of region-wide consumption and high percentages of frequency occurrence, biomass, and or number consumed), with the exception of Lake Superior (Table 11). In Lake Superior fishes from the littoral and bottom-dwelling groups, stickleback and sculpin sp., respectively, appeared to be most important (Table 12, Table 13). These species were also the second most important species consumed region-wide. Cyprinids, especially *Notropis* sp., appeared to be an important dietary component, in terms of % frequency and number, in several locales during specific time periods. Yellow perch had a fairly high frequency in the diet in several locales during specific time periods (Table 12); however, none of the studies reported significant impacts to the yellow perch fishery. In most locales, salmonids did not comprise an important portion of the diet (Table 14). Locally important species include burbot, freshwater drum and lake / northern chub (Table 15).

Diet in the Southeastern U.S.

We found four diet studies that were conducted in the 1990s that examined the diet of cormorants wintering in southeastern U.S. Most of the diet in this region consists of shad, catfish, and sunfish species (Table 16 – Table 19). At inland reservoirs in Texas, lakes in Mississippi and Alabama, and at Arkansas aquaculture facilities, shad species, namely gizzard and threadfin, appear to be the most important species in the diet, in terms of percent frequency and or percent number (Table 16). For birds around catfish farms in the Mississippi Delta, catfish were of major importance in the diet during winter months (Table 17), though shad species were of great importance in the diet of these birds in the fall (Table 16). At aquaculture facilities in central and s.e. Arkansas, catfish occurred in the diet with a surprisingly low % frequency (Table

17). However, Bivings et al. (1989) reported an increase in catfish consumption in December, and suggested that cormorant diet undergoes a seasonal shift to catfish from fall to winter, which is supported by the pattern of catfish consumption reported in the Mississippi delta (Table 17). At certain times and in certain locations, centrarchids, mainly sunfish and crappies, appear to be moderately important in the diet (Table 18). blue tilapia was locally important in Texas in winter months (Table 19).

Diet in the Atlantic Region

In the Atlantic region, the studies we reviewed indicated that diet consisted of fish and crustaceans. Six categories were developed (Cairns 1998) for prey species consumed: marine bottom fishes, estuarine / diadromous fishes, freshwater fishes, salmonids, pelagic fishes, and crustaceans (Table 20 – Table 25). Diet varied regionally to a great extent, and no one species or category emerged as the most important on a region-wide basis. However, most colonies are located on coastal islands in marine habitats, and several of the marine bottom fish species, e.g., cod, sculpin, cunner and gunnel (Table 20), were important in many areas, as were pelagic fishes such as sand lance and capelin (Table 24) (J.F. Rail, pers. comm.). In the spring, cormorants are also found on lakes, rivers and estuaries, and at this time they feed mainly on estuarine / diadromous fishes, such as alewife, rainbow smelt, and stickleback (Table 21); freshwater fishes, such as smallmouth bass, yellow perch, pumpkinseed and cyprinids (Table 22); and salmonids, mainly Atlantic salmon, especially in May (Table 23). However, this diet may not be representative of the diet of migrating DCCOs in general; use of this type of habitat may involve only relatively small numbers of individuals compared to large populations migrating / breeding in the Atlantic region (J.F. Rail, pers. comm.). Crustaceans, mostly shrimp species (Table 25), were important locally; importance of shrimp in the diet was highly variable with month and year.

Conclusions: Variation in the Diet

One of the most important features of cormorant diet that emerged from our review of diet studies was the great degree of variation that exists in utilization of prey. Table 7 – Table 25 highlight some of the variation that occurs locally, regionally, temporally, seasonally, and inter-annually. Because of this variation, measurements of diet composition must be interpreted cautiously. Neuman et al. (1997) conducted the first extensive study on the diet of DCCOs on the lower Great Lakes that explored temporal and spatial variation in detail. They suggested that, as a result of the importance of temporal and spatial variation within cormorant diet, the temporal and spatial scales at which biologists and fishery managers have previously investigated the effect of DCCO predation on fish communities may introduce considerable bias and error. In the Atlantic region, Blackwell et al. (1997) did a similarly extensive study exploring temporal and spatial variation in cormorant diet. They found that spatial and seasonal variation of prey populations had a major influence on cormorant habitat use and diet composition. On the Pacific Coast, Roby et al. (1998) studied DCCO diet in two Columbia River estuary colonies about 15 miles apart, and found that proportions of salmonids and other fish species varied significantly between locations. In general, cormorant diets show a high degree of spatial variability and tend to reflect the fish species composition for each site, making it necessary to examine cormorant diets on a site-by-site basis (Belyea et al. 1999). Additionally, cormorant diet also tends to reflect both short term and long term temporal shifts in the abundance of prey species (Blackwell et al. 1995; Neuman et al. 1997). Therefore, Neuman et al. (1997) recommend that sampling be

conducted at a scale appropriate to the question being posed, and noted that extrapolations to whole-lake context can only be made meaningful once spatial and temporal variability has been quantified.

Fish species declines associated with DCCO predation (perceived or real)

Zone 1, Alaska, is the only portion of the DCCO's breeding range where no serious conflicts between cormorants and fisheries are currently known to exist (V. Byrd, S. Stephensen and K. Kuletz, pers. comm.). In Zones 2 – 5, cormorants are associated with negative impacts to sport and commercial fish species. In the Great Lakes, fisheries managers and sport and commercial fishermen have been concerned that predation pressure from growing populations of DCCOs has or will contribute to declines in yellow perch, white perch, smallmouth bass and walleye (Craven and Lev 1987; Belyea et al. 1999; Bur et al. 1999; Matteson et al. 1999; Lantry et al. 1999; Schneider and Adams 1999). In the Atlantic region, concerns have revolved around impacts to Atlantic salmon, brook trout and rainbow smelt (Milton et al. 1995; Blackwell et al. 1997; Hill et al. 1997). On the Pacific Coast, concerns center on impacts to salmonids, particularly in the Columbia River estuary (Roby et al. 1998). In the southeastern U.S. cormorant foraging at catfish farms is reported to result in serious economic losses, and impacts to sport fishes are also of concern (Glahn et al. 1995; Campo et al. 1993; Simmonds et al. in press). At Lake Winnipegosis, Manitoba, commercial fishermen have been concerned that cormorants were negatively impacting walleye and sauger (Hobson et al. 1989).

A review of most diet literature indicates that sport and commercial fish species, on the whole, do not contribute substantially to cormorant diet and / or are not negatively impacted by cormorants (Taverner 1915; Lewis 1929, 1956, 1957; Mendall 1936; Scattergood 1950; Ross 1973; Robertson 1974; Karwowski et al. 1994; Campo et al. 1993; Hobson et al. 1989; Glahn et al. 1998; Bivings et al. 1989; Roney 1979; Craven and Lev 1987; Milton et al. 1995; Kehoe 1987; Ludwig et al. 1989; Matteson et al. 1999; Madenjian and Gabrey 1994, 1995; Diana et al. 1997 or Belyea et al. 1999; Kirsch 1995; Bur et al. 1999; Blackwell et al. 1995; Ross and Johnson 1995; Johnson and Ross 1996; Johnson et al. 1999; Ross and Johnson 1999b; Suter 1995; Weseloh and Casselman 1992; Rail and Chapdelaine 1998). There are, of course, exceptions to this finding. Birt et al. (1987) reported that fish densities were significantly lower in Prince Edward Island bays used by cormorants for feeding than in those outside their foraging range, and suggested that cormorants depleted prey in these bays; however, there is no commercial fin fishery in these bays. Cairns (1998), in his review of diet studies spanning 1915 – 1997 in the Atlantic region, stated that commercial and recreational species averaged 73 % of reported prey composition during the smolt run, and 44 % outside the smolt run. Blackwell et al. (1997) and Milton et al. (1995) found that salmon smolts figure prominently in the diet of cormorants in this region in May. However, cormorant predation on smolts in Maine is not thought to seriously impact the salmon fishery, because cormorants are only one of a myriad of mortality factors for hatchery-raised smolts (R.B. Allen, pers. comm.). In the southern U.S. catfish raised in aquaculture ponds contribute significantly to the diet of wintering cormorants (Glahn et al. 1995). Additionally, in the Great Lakes yellow perch are sometimes abundantly represented in the diet of cormorants, though significant impacts have not been documented (Matteson et al. 1999).

Studies assessing impacts to sport and / or commercial fish populations

As noted above, many studies have been conducted to examine the diet of the DCCO and to document effects of predation. Despite these efforts, there are surprisingly few studies available worldwide that include enough data for assessing the impact of bird predation, and modeling attempts are almost nonexistent (Suter 2000). However, the perception that cormorant predation is destroying fisheries is strong, particularly among anglers. To properly manage both bird and fish populations, scientifically rigorous studies that quantify the impacts of birds on fish population dynamics as well as on fisheries are needed (Suter 2000).

Such studies are difficult to undertake for several reasons. Fisheries biologists caution that documenting amounts of fish species X consumed by DCCOs does not readily translate into determining the impact on species X's population. An increase in a single predator species like the DCCO can lead to complex interactions among other predators, interspecific competitors, and prey populations that can confound the ability to draw simple conclusions about the effects of consumption rates on commercial fish population dynamics (S. Correia, pers. comm.). Trapp et al. (1999) note that to assess the impacts of DCCOs on fish populations, quantitative information about numerical abundance of prey species and detailed knowledge of effects of all other factors (both biotic and abiotic) that can affect fish populations are required. To date, no studies have been able to thoroughly address all of these criteria. However, some investigators have obtained current estimates of prey fish population size and mortality, DCCO numbers, fish consumption rates, and sport fish catch to assess potential impacts of DCCO predation. These studies are summarized below. Particular attention is paid to methodologies used to assess cormorant diet, and the scientific rigor with which estimates of fish population size and other parameters were obtained.

Great Lakes

In the Great Lakes, three studies incorporate current estimates of abundance of specific sport / commercial prey fish species to determine the impact of DCCO predation. These are reviewed below. A fourth study (Schneider et al. 1996) examined the impact of DCCO predation on smallmouth bass stocks in the eastern basin of Lake Ontario, but used historical data sets to estimate smallmouth bass abundance. Because this study utilized data sets that did not provide a measure of current conditions and made a number of assumptions from limited data, we did not review this study, but citation information is provided in the Literature Cited section. A fifth study (Hoyle et al. 1999) examined population trends and declines in smallmouth bass in eastern Lake Ontario in relation to water temperature. This study is briefly reviewed to put cormorant predation in the context of other environmental factors and ecosystem processes.

The Les Cheneaux Islands, Michigan (Belyea et al. 1999)

To determine impact of cormorants on the yellow perch population in the Les Cheneaux Islands area, MI, Belyea et al. (1999) estimated the amount of various fish taxa consumed by cormorants in 1995 through an analysis of stomach contents. Diet composition was determined by calculating the proportion by weight of each fish taxon in the diet. Weights of fish consumed were estimated by using length – weight regressions; the majority (70 %) of yellow perch found in stomachs that were measured for length were measured directly, while the remaining 30 % were estimated from cleithrum or preopercle lengths. Estimates of daily food consumption of cormorants were also calculated, based on published daily calorie intake information, and dividing it by the caloric density of fish.

Estimates of yellow perch population size and mortality rates calculated for 1995 were incorporated from Schneeberger and Scott (1997). Schneeberger and Scott (1997) estimated yellow perch population size through a mark-recapture effort conducted in 1995. Mortality was calculated for yellow perch caught in gill nets during fall 1995 and aged from scale samples. Additionally a mortality rate was calculated from a pooled data set (1993 to 1995), and a total annual mortality rate of 45 % was estimated for the yellow perch population. Survival was calculated from tag-return data. Belyea et al. (1999) then calculated yellow perch mortality from cormorant predation by using the daily caloric data together with cormorant population data and the proportion by mass of perch in the diet. They estimated that cormorants removed 1 % of the yellow perch population that was of legal size in 1995, while angler exploitation removed 2.4 %. Belyea et al. (1999) compared their cormorant predation and angler exploitation rates with Schneeberger and Scott's (1997) mortality rates, and concluded that other sources of mortality must therefore remove roughly 40 % of legal size perch.

Although cormorant diet contained approximately 48 % yellow perch by weight during the spawning season, Belyea et al. (1995) suggested that cormorant predation of perch in 1995 was, on the whole, not substantial. Additionally, Belyea et al. (1999) reported that cormorant predation on yellow perch in the Les Cheneaux Islands area accounts for a greater proportion of mortality at younger ages (fish aged 1 – 3 years), while anglers impact essentially age 3 and older perch.

Comments on Study. The estimate for yellow perch population size had a 95 % confidence level of $\pm 747,597$ (27 % of the estimated population). In calculating this estimate, there were six assumptions that were implicit to justify the formula used to estimate population size. All of these assumptions were met. Assessment of diet composition utilized estimates of fish lengths obtained from measurements of mainly intact fish, but also from potentially robust key bones (otoliths were avoided, G. Belyea, pers. comm.). While estimating diet from intact items alone has been shown to underestimate the proportion of small fish species and overestimate the average size for some larger ones (see above, Diet Assessment Methods: Advantages and Shortcomings), care was taken to collect birds shortly after they had been observed feeding. This was done to minimize the number of birds shot with empty stomachs; it may also have increased chances that most fish retrieved from stomachs, even smaller ones, would be relatively undigested. Estimates of daily food consumption incorporated published daily calorie intake information determined both observationally and experimentally. While the estimate for population size had fairly wide confidence intervals, given the level of yellow perch mortality due to cormorant predation, it was precise enough to conclude with some certainty that DCCOs were not substantially impacting the yellow perch fishery.

Eastern Basin of Lake Ontario (NYSDEC 1999)

The New York Department of Environmental Conservation (NYSDEC) and United States Geological Survey (USGS) undertook an intense field effort in 1998 to evaluate the impact of DCCO predation on smallmouth bass and other fishes of the eastern basin of Lake Ontario (NYSDEC 1999). The area specifically studied included New York waters of Lake Ontario from 7 km south of Little Galloo Island to Cape Vincent, and throughout this review will be referred to as the eastern basin. The document produced from this work contains 12 papers addressing various aspects of the warm water fish community and DCCO diet in this area. Some of these papers are summarized here to present the major results of this effort, and impacts of DCCO predation in this area.

In the eastern basin, annual assessments of the warm water fish community, based on gill net sampling conducted by the NYSDEC, have reported dramatic declines in smallmouth bass abundance since the early 1990s. The mean catch per net gang (or catch per unit effort, CPUE) from 1995 – 1997 was more than 50 % lower than the mean CPUE from 1984 – 1986 (Chrisman and Eckert 1999). Additionally, Eckert (1999a) reported a decrease in annual survival rate among bass ages 6 – 12 since 1991, and a progressive shift in the modal age group of bass captured towards younger age 3 and age 4 fish, suggesting decreased survival among younger age classes as well. Because increases in DCCOs have corresponded with these declines, efforts have been made to determine cormorant predation impacts on smallmouth bass abundance.

Johnson et al. (1999) assessed the diet of cormorants in eastern Lake Ontario by collecting pellets from Little Galloo Island and identifying diagnostic structures, primarily fish otoliths. They then developed a model to estimate numbers of fish (for each prey species) consumed by cormorants in eastern Lake Ontario. One important assumption of the model was that cormorant's produced 1.0 pellets per day. Model results estimated that 1.3 million smallmouth bass (the species of greatest concern) were consumed in 1998.

To evaluate the size and age structure of smallmouth bass being consumed by cormorants, two studies were carried out: Adams et al. (1999) utilized otoliths from cormorant pellets collected in 1993 – 1994, and Schneider and Adams (1999) utilized otoliths and other fish parts from cormorant pellets, chick regurgitations and stomach contents collected in 1998. To minimize potential bias associated with otoliths of consumed fish, an erosion index scale was developed in which otoliths were assigned to various categories by degree of wear; only the least eroded and most reliable predictors of fish lengths were used to estimate total length of fish consumed (Adams et al. 1999; Schneider and Adams 1999). These studies concluded that the smallmouth bass consumed by cormorants were mostly juveniles, between the ages of 3 – 5. Adams et al. (1999) reported that the modal age group of bass consumed in 1993 – 1994 was 5 years; the mean age was 4.4 years. Schneider and Adams (1999) reported that the modal age of bass consumed in 1998 was 3 years; the mean age ranged from 2.7 – 3.3 years for the three sample types.

The objectives of the final paper in the NYSDEC's Special Report (Lantry et al. 1999) were to determine if cormorant predation significantly increased mortality of smallmouth bass in the eastern basin, and if this mortality could cause the declines recorded in the smallmouth bass population and fishery. Lantry et al. (1999) synthesized research from several sources. The percent of smallmouth bass in the diet was based on the estimate by Johnson et al. (1999). The estimate of smallmouth bass abundance and density was based on a 1986 mark-recapture population estimate for adult smallmouth bass in the Hardscrabble area (McCullough 1988) and expanded to yield a density estimate for the area in the eastern basin between the 0 to 20 m depth contours. The population estimate was back-projected for ages 1 to 5 using survivorship values for young smallmouth bass obtained at Oneida Lake in the 1970s (Carlander 1977). Age distribution of smallmouth bass preyed upon by cormorants was obtained from Adams et al. (1999) and Schneider and Adams (1999). Lantry et al. (1999) then calculated the percent of 3 – 5 year old bass preyed upon by cormorants in the eastern basin, and estimated that cormorant predation reduced the 3 – 5 year old age group by 23 % in 1993 – 94, and by 36 % in 1998. The main results reported by Lantry et al. (1999) were: 1) the mortality of age 3 to 5 smallmouth bass increased substantially after 1988; and 2) loss rates of the magnitude reported above could severely limit numbers of adult stock recruited to the fishery; and 3) cormorant predation on 3 –

5 year old age classes was substantial enough to cause the observed declines in the smallmouth bass population.

Eckert (1999b) also identified cormorant predation as one of the factors that may have decreased yellow perch survival between the 1970s – 1990s in the eastern basin of Lake Ontario. However, more quantitative information is necessary to assess impacts of cormorant predation on yellow perch in this area.

Comments on Study. This combined study synthesized a wide range of both cormorant and fisheries data. The DCCO diet sampling was especially thorough, covering a broad temporal range, utilizing a very large sample size, and combining various assessment methods to determine diet composition. Age estimates for consumed fish were included, and efforts were made to assess large scale and long-term trends in fish populations. Utilization of such a database and the broad perspective brought to the cormorant-fishery interaction in the eastern basin sets an important direction for future research. However, a few of the methodologies, assumptions and estimates incorporated in the individual papers have associated potential biases that complicate interpretation of results. These are noted below.

The conclusion that cormorant predation on 3 – 5 year old smallmouth bass age classes was substantial enough to cause the observed declines in the smallmouth bass population rested largely on the age distribution of smallmouth bass consumed by cormorants, which was obtained through analysis of otoliths retrieved from pellets. Several researchers have questioned this method of aging consumed fish (see above, Diet Assessment Methods: Advantages and Shortcomings). However, attempts to minimize such biases were made by incorporating an erosion index scale for the otoliths used.

The assumption that cormorants produce 1.0 pellets per day may be an inaccurate rate of pellet production, and as such may have led to an over- or underestimate of the number of fish consumed. Recent field research on the rate of pellet formation for other Phalacrocoracidae species suggests that this is too high a rate of pellet production (see above, Diet Assessment Methods: Advantages and Shortcomings).

The estimate for smallmouth bass population size and density was calculated in 1986 and expanded to the area in the eastern basin between the 0 – 20 m depth contours to produce a current smallmouth bass density estimate. However, in 1986, numbers of smallmouth bass appeared to be much higher than those in 1998. Given the uncertainty of this model input, it is not clear how much confidence can be ascribed to this density estimate.

Survival rates for young smallmouth bass were based on rates from Oneida Lake. This is a much smaller inland lake, subject to a different set of ecosystem dynamics. It is not clear how applicable these survival rates are to smallmouth bass in the eastern basin.

An important question not addressed in this report is that there were periods when smallmouth bass declines corresponded with low numbers of cormorants, and periods when both species increased simultaneously. For example, between 1980 – 1985 smallmouth bass abundance (measured by CPUE) declined by about 75 % (Chrisman and Eckert 1999); at this time cormorant numbers on Little Galloo were relatively small (< 1000 pairs up until 1985, when about 1400 pairs were estimated). Then during 1985 – 1990, smallmouth bass abundance dramatically increased (Chrisman and Eckert 1999), and cormorant numbers nearly tripled. This suggests that other factors might be responsible for smallmouth bass declines. Potential factors such as disease, decreased food supply, changing water quality conditions in the 1990s, zebra mussel alterations on habitat, fish predators and other bird predators have not been thoroughly explored or quantified.

Diet Studies at Oneida Lake, New York

(Forney 1993; VanDeValk et al. 1998, 1999; C. Adams, pers. comm.)

At Oneida Lake, New York, various aspects of DCCO diet and interactions between cormorants, anglers, walleye and yellow perch have been studied since 1988 (Forney 1993), but no peer-reviewed papers describing the study have been published. Therefore we have summarized work conducted there through communications with the researchers involved in the work (Connie Adams and Lars Rudstam), descriptions in annual reports prepared by VanDeValk et al. (1998, 1999), and through observations on cormorant – fish interactions published in the American Fisheries Society Newsletter (Forney 1993). On Oneida Lake, estimates of abundance, age composition and survival for yellow perch and walleye were obtained through Miller sampler surveys, gillnet catches, catch in trawls, and mark-recapture studies (VanDeValk et al. 1998, 1999). Cormorant diet was examined 1988 – 1991 through analysis of regurgitations and stomach contents (Forney 1993); 1994 – 1997 through analysis of pellets, regurgitations and stomach contents (VanDeValk et al. 1999; C. Adams, pers. comm.); in 1998 and 1999, diet was examined through regurgitants (C. Adams, pers. comm.). Cormorant diet composition was described by % number and % biomass. Numbers of walleye and perch consumed by cormorants incorporated daily consumption rates of adult cormorants and chicks, proportion by weight of different prey items in the diet, and the number of adults and chicks observed on the lake (VanDeValk et al. 1998). Daily consumption was estimated from two methods. In the first, it was calculated as the average number of fish found per pellet, and assumed that cormorants produced 1.0 pellets / day. In the second method, it was assumed that adult cormorants consume 20 % of their body weight per day and that chicks consume 0.327 g / day / individual. [Chick consumption rates were based on bioenergetics calculations from Fowle et al. 1997 (C. Adams, pers. comm.).] Proportion of different age classes consumed by cormorants was estimated through analysis of fish remains obtained in pellets, regurgitants and stomach contents. As in the 1999 NYSDEC study, efforts were made to minimize potential bias associated with aging consumed fish through otolith measurements by incorporating an erosion index scale (C. Adams, pers. comm.). Exploitation rates of yellow perch and walleye were compared between cormorants and anglers. Angler take was estimated through creel surveys (C. Adams, pers. comm.).

For both walleye and perch, fishery biologists at Cornell Field Station use the number of age 1 fish to predict the number that will be recruited to the adult population. Predation at this level and on the pre-recruit populations (age 2 – 3) will have the greatest effect on numbers recruited to the fishery, and is believed to represent additive mortality. Essentially, predation on age 0 walleye and yellow perch has no impact on the resulting number of adult fish (age at recruitment: age 4 for walleye, age 3 for yellow perch). Because it is only from age 1 that the number of adults from a year class can be predicted with some confidence, compensatory effects are believed to be less pronounced after age 1. Therefore DCCO predation on these larger fish will directly decrease the number of adult fish recruiting from that year class (C. Adams and L. Rudstam, pers. comm.). In the spring of 1996, a total of 8,821,000 yellow perch aged 1 and older was estimated. The amount consumed by cormorants was estimated at 1,123,800 (12.7 %). A total of 697,935 walleye aged 1 and older was estimated. The amount consumed by cormorants was estimated at 100,600 (14.4 %) (VanDeValk et al. 1999). Creel surveys and pellet analyses conducted in 1997 – 1998 indicated that exploitation of adult walleye was solely attributed to anglers while adult yellow perch exploitation was divided about evenly between anglers and

cormorants. Exploitation of subadult walleye and yellow perch was almost entirely attributed to cormorant predation (C. Adams, pers. comm.).

Before cormorants colonized Lake Oneida, the relationship between number of age 1 fish and resulting number of adults was tightly correlated. Since the growth of the cormorant population, this equation consistently predicts more adults than are estimated with mark-recapture counts, suggesting an increase in mortality between age 1 and age at recruitment.

When the number of fish eaten by cormorants is calculated, it is a fair percent of the number of fish that are “missing.” Therefore, depredation by cormorants is believed to have an impact, as virtually the only other cause of mortality for adult walleye and yellow perch is due to angling (C. Adams, pers. comm.).

After reviewing cormorant - fish interactions between 1988 – 1991, Forney (1993) reported that cormorant predation does not seriously threaten the Oneida Lake fishery, but noted that a continued increase in cormorant abundance may intensify demands for cormorant control. The last year of data considered by Forney (1993) was 1991, when 60 DCCO pairs were estimated to nest on the lake, and flocks of 300 – 400 fall migrants were common; however, for some years, 1,000 – 2,000 cormorants were reported. Since 1991, nesting numbers have increased, with 332 nests counted in May of 1998, though through control efforts only 100 were allowed to persist. In the same year, about 1,500 birds utilized the lake in August, but quickly declined by the end of September to 20 birds following the onset of harassment (VanDeValk et al. 1999). The more recent work conducted on Oneida Lake bases a conclusion of effect on perch and walleye on three lines of evidence: 1) the timing of the disappearing adults (fewer adults than expected from age 1 estimates) coincides with the increase in cormorants; 2) the size of the fish eaten by cormorants coincides with the size of the fish that have increased mortality; and 3) for walleye at least, the number of fish “missing” is comparable to the number estimated consumed by cormorants (C. Adams and L. Rudstam, pers. comm.).

Comments on Study. The work done on Oneida Lake stands out overall in its examination and discussion of compensatory and additive processes within fish populations and provides some of the best evidence for DCCO impacts on fish populations. A variety of estimates for numbers of walleye and yellow perch have been incorporated in an effort to provide as complete a picture of fish populations as possible. Assessment of cormorant diet was thorough and utilized a combination of methods. Many of the fish consumed by cormorants were sized and aged by samples obtained through regurgitants and stomach contents. A few potential limitations of the data are noted below.

The population estimates for walleye and perch were current and carefully obtained through mark-recapture and other frequently utilized methods. The estimate for walleye had narrower confidence intervals than the estimate for yellow perch; however, one of the two assumptions discussed for the model used for the walleye estimate was not met. The population estimate for yellow perch had sizable confidence intervals because only a small number of individuals was recaptured; assumptions for this model were not discussed. Additionally, the numbers of walleye and perch taken by cormorants fell largely or completely within these intervals (VanDeValk et al. 1998, 1999).

Otoliths retrieved from pellets were used to age some of the consumed fish. For perch the researchers believed that the age classes of consumed fish were determined fairly accurately, as the otolith size distribution exhibited distinct peaks that could be identified as age 0, 1, 2 and 3 + fish. For walleye too few otoliths were measurable, and thus the size of walleye was calculated from the size of the otolith using species-specific regressions, which resulted in an underestimate

of walleye consumed due to erosion of the otoliths (C. Adams, pers. comm.). It is likely that small otoliths were lost, and thus consumption of younger walleye was probably also underestimated. Additionally, the authors assumed that 1 pellet per day accurately reflects DCCO daily food intake, which may lead to under- or over-estimation of consumption (see Diet Assessment Methods: Advantages and Shortcomings, and review of Eastern Basin of Lake Ontario (NYSDEC special report 1999).

Modeling predicts that even elimination of the entire cormorant population on Oneida Lake would not lead to recovery of the walleye population to its formerly large size. The population increases, but then levels off at a fairly low number. Because it does not continue to increase, there must be other factors besides cormorant predation holding it at lower levels. One element in the model that causes a rapid and steady increase in the walleye population is closing the fishery to angler harvest. It appears that if this were done for several years, the walleye population would eventually reach the high population of the 1950s. Anglers remain the most important predators of walleye on Oneida Lake (C. Adams, pers. comm.). Additionally, other recent changes observed in the Oneida Lake ecosystem may also affect recovery of the walleye population, including declining phosphorous and other nutrient levels, and invasion of zebra mussels with the associated change in water clarity and decline in algal concentrations (VanDeValk et al. 1999). In terms of the latter effect, the number of young of year walleye caught in the index trawl has been consistently low since 1992, suggesting higher early mortality, which could be increasing due to observed higher water clarity. In turn, this would increase susceptibility of this age class to predation / cannibalism. If this trend could be reversed, higher numbers of young walleye could compensate for the apparent increase in mortality in walleye aged 1 – 4 (VanDeValk et al. 1999). However, we do not know if factors such as these were included in modeling efforts.

Eastern Lake Ontario, Ontario, Canada (Hoyle et al. 1998)

Smallmouth bass abundance in the Canadian waters of the eastern basin of Lake Ontario was monitored using gillnets from 1978 to 1998. Age of caught bass was interpreted from scales. Annual survival rate was estimated by year-class for fish aged 6 to 10 years old. Smallmouth bass year-class strength was estimated by two methods. Age-specific CUEs were summed for fish aged 2, 3 and 4 years old by year-class, and provided complete year-class strength information for 17 year-classes from 1978 to 1994. Year-class strength was also estimated for ages 2 to 11, using proportional year-class strength. Proportional year-class strength for 23 year-classes from 1973 to 1975 was calculated. Year-class strength estimates were then correlated with July / August water temperature data. For water temperatures, two databases were used.

Hoyle et al. (1999) concluded that smallmouth bass year-class strength, as influenced primarily by midsummer water temperatures, appeared to be the major factor determining the size of the smallmouth bass population in the eastern basin of Lake Ontario between 1978 – 1998. Cumulative gillnet CPUEs for ages 2 to 4 years showed a strong year class in 1995, and based on year class strength estimates for the years 1995 to 1998, Hoyle et al. (1999) predicted that index gillnet catches should increase over the next few years.

Hoyle et al. (1999) comment on three other factors that may affect size and trends of the smallmouth bass population. They state that the annual survival rate for smallmouth bass aged 6 to 10 years old (mean = 0.62) suggests that overexploitation is not a problem, but they also note that one explanation consistent with the observation that survival of strong year-classes (mean = 0.54) is lower than that for weak year-classes (mean = 0.67) may be that strong year-classes were “fished-up” (i.e., exploitation rates were highest in years when bass were most abundant). Noting

that cormorant predation has been implicated as influencing survival of young smallmouth bass in New York waters of eastern Lake Ontario, Hoyle et al. (1999) comment that if cormorants are a major source of smallmouth bass mortality in eastern Lake Ontario, their impact has been too recent to distinguish it from expected population trends due to the recent history of summer water temperature and low year-class strength. In conclusion, they note that significant ecosystem changes in eastern Lake Ontario have occurred since the arrival of Dreissenid mussels (*Dreissena sp.*) in the early 1990s, and state that associated changes (increased water clarity, fundamental shifts in food-web interactions) are certain to impact smallmouth bass distribution and abundance at some level.

Comments on study. Estimates of smallmouth bass abundance, survival and age distribution were calculated through standard methods. These estimates appeared rigorous enough to detect population trends over the 20 year period examined, and to document declines in abundance similar to those reported for smallmouth bass in New York waters of the eastern basin during about the same time period (Chrisman and Eckert 1999). Viewing these population trends in relationship to water temperature provides an example of how abiotic factors may influence population size, and the importance of examining a variety of factors in efforts to determine factors contributing to fish population declines.

Southeastern U.S.

In the southeastern U.S., two studies have been undertaken that incorporate some of the data necessary to provide information on impacts to fish populations. The study by Glahn et al. (1998) incorporates estimates of abundance of specific sport fish species and DCCOs to determine the impact of DCCO predation (Glahn et al. 1998). The study by Simmonds et al. (in press) presents a model to evaluate effects of DCCO predation on standing crops and yields of reservoir fish species and incorporates various parameters.

Lake Beulah, Mississippi, and Lake Eufala, Alabama (Glahn et al. 1998)

Glahn et al. (1998) compared percentages of prey species in the diet of cormorants at Lake Beulah, MS, and Lake Eufala, AL, to the percent availability of these prey species in the lakes. Fish availability at Lake Beulah was based on gill net, trammel net and hoop net sampling in 1995; availability at Lake Eufala was based on gill net sampling in 1997. Analysis of stomach contents was utilized to determine cormorant diet composition. For birds collected at Lake Beulah, length measurements of intact and partially digested fish were used in species-specific length-to-weight equations to obtain % biomass of each species in the diet. For birds collected at Lake Eufala, few intact fish were found in stomachs, and otolith analysis was the primary means used to identify prey species and number of fish consumed. To avoid problems associated with possible otolith erosion during digestion, fish from Lake Eufala were aged by counting hyaline and adjacent opaque bands of sagittal otoliths. From these age classifications median fish length was assessed from previous length-frequency data from Lake Beulah. The number of shad and bluegill removed from Lake Beulah by cormorants over a 4-month period was also estimated, based on average number of cormorants observed foraging on the lake, fish species composition by weight of the diet, and average daily cormorant food demand (estimated previously, Glahn and Brugger 1995).

The only sport fishes that occurred in the diet in substantial numbers were sunfish species, particularly bluegill. Glahn et al (1998) estimated that cormorants consumed only a small percent of bluegill available in Lake Beulah, but acknowledged that their estimate of bluegill consumed may have been low. They concluded that, with the possible exception of

predation on harvestable size bluegill, cormorants do not appear to have an appreciably negative impact on sport fisheries at the two lakes studied.

Comments on Study. Assessment of diet composition utilized estimates of fish lengths obtained directly from measurements of intact fish, partially digested fish, and indirectly, by aging fish through counting hyaline and opaque bands. Estimates of daily food consumption were based on an earlier study that carefully calculated average daily fish consumption based on bioenergetics data. Estimates of fish abundance in both lakes utilized standard methods; however, inconsistencies in results and fish sampling techniques that were possibly biased against capturing bluegill relative to other species prevented the authors from concluding definitively that cormorants had no appreciably negative effect on sport fish. The apparent preference cormorants had for bluegill may have been overstated, though bluegill and other sunfishes appeared important in the diet at both lakes.

Because sizes of fish consumed by cormorants in the two lakes were estimated with two different methods, there may have been inconsistencies in the results; selectively collecting birds observed feeding at Lake Eufala might have reduced the number of birds without intact fish in their stomachs, and enabled comparative measurements. Based on the average size of intact bluegill consumed at Lake Beulah, cormorants would be in direct competition with anglers. However, the estimated size distribution of bluegill from otolith aging suggested that < half of the bluegill consumed by cormorants were of harvestable size. From their analysis of intact fish at Lake Beulah, the authors estimated that the total number of bluegill consumed by cormorants during the winter months was a little more than 100,000. However, the total number consumed may be larger if the average size of bluegill, as estimated from their otolith analysis, was smaller. The authors note that determining the impact of cormorant predation on bluegills overall is difficult because of the wide variation in bluegill population structure between years; additional studies involving intensive sampling of bluegill populations concurrent with cormorant diet studies need to be undertaken to determine the impact from cormorant predation more conclusively.

South-central Great Plains (Simmonds et al. in press)

Simmonds et al. (in press) developed mathematical models to evaluate effects of DCCO predation on fish abundances and angling yields in a hypothetical south-central Great Plains Reservoir. Specifically, an adaptation of the Ricker equilibrium-yield model (Ricker 1975) was used to evaluate effects of DCCO predation on standing crops and yields of individual reservoir fish species. Parameters incorporated into the model included: rates of fish growth; instantaneous rates of natural fish mortality; instantaneous rates of angling mortality; DCCO induced fish mortality; DCCO abundance; and the weight of each fish species consumed per hectare per cormorant-use-day. DCCO densities and periodicities of seasonal occurrence were based on actual counts conducted at eight reservoirs in Oklahoma during two field seasons, for a total of 16 reservoir observations. Densities were highly variable among reservoirs and between years, but at most reservoirs densities were low. In 14 of the 16 observations, cormorants were present only during fall and spring; in the remaining two observations they were abundant throughout the migration and wintering periods.

Their analyses suggested that DCCO predation has only a minor or inconsequential impact on the recreational fishery of a typical (e.g., low cormorant density) reservoir in Oklahoma. At the two reservoirs where high densities of cormorants remained throughout the migration and wintering periods, impacts were more severe. However, high predation rates did not persist at either reservoir, as high densities of cormorants were only estimated in the first

year of the study; in the second year, substantially lower cormorant densities were observed. The authors suggested that long-term monitoring of DCCO abundances and temporal use patterns is needed to understand predation dynamics and comprehensively model and evaluate impacts on reservoir sport fishes in atypical (e.g. high cormorant density) Oklahoma reservoirs. Fisheries biologists have drawn similar conclusions, and suggest that impact studies should include examination of long-term trends in DCCO abundance, as detecting relations through correlation type analyses will require long time-series (S. Correia, pers. comm.).

Simmonds et al. (in press) also noted that effects of DCCO predation may be detrimental, beneficial or neutral in reservoirs if effects negate each other. Results of their model indicated that DCCO predation reduced standing crops of channel catfish, largemouth bass and white crappie, and reduced their yields to the sport fishery; but in many cases density-dependent fish growth responses largely mitigated these effects. Thus, while predation may reduce survival, standing crop, and yield of a fish population, and consumption of small fish could reduce the forage available to large sport fish, it may also decrease competition and thus increase the growth rate of the remaining sport fish. Additionally, cormorant predation could conceivably function as a biological control of gizzard shad populations, which may lead to increased availability of juvenile shad to predatory sport fish.

Comments on Study. This study did not actually assess cormorant diet or impacts to an actual fishery. However, it was included because it provided information on predation impacts in reservoirs and because it illustrates the utility of modeling to assess cormorant impacts. It also incorporated many important estimates (e.g. rate of fish growth, etc.) that most impact studies have not included, but are necessary to enhance understanding of how compensatory responses of fish populations may buffer predation effects.

Conclusions: Impacts to Sport and / or Commercial Fisheries

To summarize, three of the five studies assessing impacts to sport and commercial fisheries reported cormorants did not appear to appreciably impact fisheries (Belyea et al. 1999; Glahn et al. 1998; Simmonds et al., in press). Two reported cormorants cause significant impacts or declines within a fishery (see Oneida Lake studies above, NYSDEC special report 1999).

Review of these studies highlights the importance of several key pieces of data. One parameter that has been difficult to obtain with a high degree of precision is an estimate of prey fish population size. This is an essential estimate for determining impact from DCCO predation, because it is required to calculate the proportion of the prey fish population consumed by cormorants. Another important parameter is an accurate age class distribution of fish consumed. This parameter is needed because predation on certain age groups is likely to have a much greater effect on recruitment to a fish population and the fishery's subsequent size than would predation on other age groups (e.g., predation on 1 – 3 year old vs. young of the year walleye). Studies to determine age distribution of cormorant prey necessarily rely on examination of fish parts that have been subjected to the highly acidic environment of the DCCO's stomach. For an accurate age distribution of fish consumed, and thus better assessment of impacts to a fishery, methods of diet assessment that have less associated bias should be utilized whenever possible. In some cases this may mean relying on a smaller sample size, for example, if fish parts are retrieved from stomach contents as opposed to pellets. In some situations, a smaller sample of birds with stomachs containing fresh food (n = 12 – 15) may be adequate (Carss et al. 1997).

A fundamental question that studies need to address is that of compensatory vs. additive mortality within a fish population due to cormorant predation (Hatch and Weseloh 1999).

Interactions among mortality factors are not well understood, and releasing fish from the predation pressure of one predator (e.g. cormorants), may not necessarily lead to higher catches of fish, because these fish may be lost to other causes (e.g. other predators, poor growth, higher mortality associated with high fish density, poor overwinter survival) (Draulans 1987; Bayer 1989; Marquiss and Carss 1997; Carss et al. 1997; Derby and Lovvorn 1997b). Forney (1993) noted that “removal of walleye by cormorants could increase survival of perch” on Oneida Lake because yellow perch under 200 mm are vulnerable to predation by piscivorous fish and mortality is often high. Thus removal of cormorants from this complex system could potentially lead to further declines in perch.

Additional information is needed on compensatory mechanisms within fish populations that may operate to buffer effects of predation (Carss and Marquiss 1997, 1999; Marquiss et al. 1998; Rose et al 1999). If fish populations exhibit compensatory responses to predation, and mortality due to cormorant predation may not be additive, measuring cormorant predation relative to fish abundance may still not allow quantification of cormorant contribution to decline in fish stocks. Therefore cormorant predation should be examined in the context of other mortality factors.

Overall, comprehensive studies within systems to determine net effect of cormorants and the nature and magnitude of cormorant predation have not been conducted, primarily because of the difficulties involved in acquiring several key pieces of data. Without better understanding of fish population dynamics and responses to predation, it will be difficult, or impossible, to assess predation impacts, even when we know the mortality rate of fish due to DCCO predation. Studies such as those conducted on Oneida Lake and the work by Simmonds et al. (in press) have incorporated, or are incorporating, some data that attempt to deal with these mechanisms.

In conclusion, rigorously demonstrating whether or not DCCOs negatively impact fisheries requires precise estimates of several parameters by both fisheries and cormorant biologists. Of the studies reviewed, only one (Belyea et al. 1999) obtained estimates rigorous enough to determine impacts of DCCO predation. If this study had found that mortality of the adult perch population due to DCCO predation was considerably higher than the estimated 1 %, more precise data on fish population size and compensatory processes within the yellow perch population would perhaps be necessary to determine the impact such a predation rate has on both fish population size and subsequent catch. This observation suggests that the data required to determine impacts will vary to some degree depending on the unique conditions of the aquatic ecosystem under study. Depending on the results of initial studies, further sampling may be needed.

Studies assessing impacts to aquaculture

Most studies to assess aquacultural impacts due to cormorant predation have been conducted in the southern U.S., especially in the Mississippi Delta region, which is the center of the catfish production industry. Attempts to assess these impacts have utilized a variety of techniques: 1) surveys of catfish farmers have been conducted to assess economic losses; 2) studies have collected diet data, observed cormorant predation at ponds, and calculated cormorant predation rates; and 3) bioenergetics models have been developed to predict cormorant predation impacts. While some studies reviewed do not directly assess impacts, they were included because they provide relevant information on perception of impacts, and on aspects of cormorant foraging ecology that may result in spatial, temporal and sexual variation in levels of impact.

Surveys

Catfish Producers in the U.S. (Wywialowski, 1999)

In January of 1997, the National Agricultural Statistics Service (NASS) surveyed catfish producers in the U.S. about wildlife caused losses in 1996 (Wywialowski, 1999). Data were analyzed for six regions that included 15 states: CA, KS, MO, OK, TX, LA, MS, AL, AR, KY, TN, NC, SC, GA, FL. Fourteen hundred and sixty-five catfish producers were contacted; of these, 1,008 completed surveys. Proportionally, the greatest majority of producers were in Mississippi (n = 300), followed by Alabama (n = 163) and then Arkansas (n = 117). Producers estimated that wildlife-caused losses cost them \$12 million in lost production or repair of facilities. Additionally, about 68 % of catfish producers spent time and money (nationwide total of > \$5million) trying to avoid wildlife-caused losses of their catfish. In 1996, the total value of losses was about 4 % of the total value of catfish sales, which was \$424 million. The 4 % cost of wildlife may represent 1/3 – 1/6 of profits.

The main problem caused by wildlife, primarily birds, was feeding on catfish; the DCCO was cited most frequently as the primary species causing losses; next most frequently cited birds were herons and egrets. No breakdown of the percent of total losses caused by each wildlife species was provided. However, Wywialowski (1999) concluded that cormorant-caused losses could be anticipated to be a major problem in the Mississippi Delta region, given the growth of catfish production in conjunction with growing numbers of DCCOs that winter in the region.

Comments on study: This survey gathered important information on the percentage of catfish farmers citing wildlife-caused losses to their stocks, efforts of these producers to prevent wildlife-caused losses, and the total cost of sustained losses farmers attributed to wildlife. The survey also provided relevant information on the utilization of Wildlife Services by catfish farmers and the effectiveness of Wildlife Services in helping farmers prevent wildlife-caused losses. An important question raised in discussion of survey results was whether producers can actually identify the species and amount of losses caused by wildlife, but the survey did not incorporate questions to specifically address these questions.

Catfish Producers in Mississippi (Stickley and Andrews 1989)

In the late 1980s, the USDA/APHIS Science and Technology Research Station at Mississippi State University conducted a telephone survey of 281 Mississippi catfish farmers about the impact of fish-eating birds on their ponds (Stickley and Andrews 1989). Farmers were questioned regarding methods used to repel fish eating birds from their ponds and their perceptions of the effectiveness and costs of harassment. Eighty-seven percent of farmers felt that fish-eating birds caused problems at their farms and warranted harassment. These farmers estimated that they spent an average of 2.6 man-hours per day harassing birds at an average annual cost of \$7400; the total annual cost of bird harassment according to these farmers was \$2.1 million. Losses to birds approximated 3 % of total catfish sales.

Moderate to heavy cormorant activity (at least 25 cormorants per day) was reported by 57 % of Delta farmers, and by 14 % of non-Delta farmers. The average cost per day for cormorant harassment statewide was \$26.00, which translates into approximately \$4700 per farmer for the six month period cormorants are present in the state. In addition to reporting survey results, Stickley and Andrews (1989) estimated that 35,000 cormorants were present in the Delta in 1988 from November through April. Based on a conservative estimate of 0.67 lb of fish / day /

cormorant at the market price at that time of \$0.78 per pound, they estimated that the value of fish lost to cormorants in Mississippi in 1988 was roughly \$3.3 million.

Comments on Study. Stickley and Andrews (1989) acknowledge that their estimate assumes that cormorant diet consists entirely of catfish, while the diet of cormorants in this region is known to include other fish, such as shad and bream (Glahn et al. 1995; Glahn and Brugger 1995). Additionally, peak numbers of cormorants in the Mississippi Delta do not remain constant over winter months as Stickley and Andrews (1989) assume, but gradually increase, reaching their peak in February or March (Glahn and Brugger 1995). Furthermore, costs were estimated based on value of fish at time of consumption (replacement value), which does not necessarily reflect losses to net harvest because of compensatory mechanisms acting within catfish ponds. Thus, even though Stickley and Andrews (1989) utilize a conservative estimate for daily fish consumption, their assumptions about cormorant diet and numbers over the winter likely overestimated the economic impact of cormorant predation to catfish farmers (Glahn and Brugger 1995; see review, below).

General Aquaculture in Florida (Brugger 1995)

Brugger (1995) conducted telephone interviews with 38 biologists, extension agents and growers in Florida to identify issues of concern and reported that currently DCCOs are not obvious pests to aquaculture; overall, little conflict occurs between cormorants and the food-fish and game-fish industries in Florida waters. Conflicts occur mainly when wintering cormorants feed at inland hatcheries, grow-out ponds, and ornamental fish ponds. Losses to food fish, primarily catfish, can be locally severe. Several game fish hatcheries have also sustained high losses of fingerlings and stockers in recent years. Cormorants may also cause damages to the tropical fish industry near Tampa Bay, but the impact of vertebrate predation in general on the tropical fish industry is not known. Because ponds are small in acreage, exclusion, scare tactics, or a change in cultural practices were suggested as potentially appropriate methods to reduce fish losses to piscivorous birds.

Comments on study: At the time this study was conducted (early 1990s) there was little conflict occurring between cormorants and fisheries in Florida waters. In addition to the survey, this study reviewed breeding and wintering population trends. Precise estimates for Florida's breeding and wintering populations are not available. If conflicts in Florida become more significant in the future, better estimates of numbers of birds will be needed.

Diet Studies and Observations at Ponds

Mississippi Delta Foraging Behavior Observations (Stickley et al. 1992)

Stickley et al. (1992) conducted an observational study in the Mississippi Delta to determine the impact of cormorant depredations on commercial catfish populations. They selected 16 different pond complexes for cormorant depredation surveys from December 1989 – April 1990. Complexes were selected on the basis of growers' expectations of depredations; within the complexes, ponds were identified that were most likely to receive cormorant use. During the surveys, observers kept a minute-by-minute count of the number of cormorants on the pond. The number and species of fish seen in the bills of cormorants on the pond surface were recorded. Approximate lengths of each catfish (based on visual observation) were also recorded. Fish catch rates were determined, and feeding rates on a "cormorant-hour" basis were summarized [e.g. 10 cormorant-hours (600 cormorant-minutes) would equate to 100 cormorants on a pond for six minutes, or 20 cormorants on a pond for 30 minutes, etc.]. The average

consumption rate for all ponds surveyed was five catfish per cormorant-hour; rates ranged from 0 – 28 catfish per cormorant-hour. The average estimated length of catfish consumed was 12 cm. Stickley et al. (1992) concluded that, at their highest feeding rates, cormorants could have a devastating impact on catfish fingerling populations over time, which in turn would result in great financial impacts.

Comments on Study. Feeding rates were estimated for selected ponds based on expected levels of high cormorant predation, and ponds experiencing low levels or no predation were subsequently excluded from the study. Thus feeding rates were not provided for ponds in general. Individual ponds appear to experience various levels of predation, and some seem to be much more attractive to cormorants than others. For example, Hodges (1989) conducted 27 weekly surveys at 50 catfish ponds in Humphreys County, MS, one of the largest catfish production areas in the country. During these surveys, cormorants were observed at only nine of the 50 ponds, and on only 14 occasions. Hodges found that “cormorants were present more often at large defunct ponds with large maximum fish size...” Therefore the predation rates calculated by Stickley et al. (1992) should not be extrapolated to ponds in general.

The feeding rate was reported in numbers of catfish per cormorant-hour, and was based on observations of entire flocks, in which an interchange of numerous birds throughout the observation period was possible. Individual birds were not tracked. Because of the manner in which the data were collected, these rates should not be applied to individual birds for extended periods of time. Additionally, care should be taken in how these rates are cited. For example, Glahn and Stickley (1995) cite the average rate, stating that, “this rate would equate to a loss of \$400 US during a 9-h foraging day for a flock of 100 birds.” Mott and Boyd (1995) make a similar statement. While the researchers did not actually apply this rate to 100 individual birds assumed to be feeding for nine hours, the way they report their calculations based on this rate gives the impression that individual cormorants may feed at this rate throughout an entire day. This is an important distinction because if individual birds did forage at these rates for nine hours, they would consume many more fish than necessary for meeting metabolic requirements and have a much greater impact than actually occurs.

Finally, this study provides estimates of economic impact based on replacement costs for fingerlings at the time of predation. However, the degree to which cormorant predation on catfish fingerlings represents additive vs. compensatory mortality is not known. Therefore, it is not possible to extrapolate these replacement costs to net impacts at time of harvest, as the impact of cormorants relative to other mortality factors has not been determined.

Mississippi Delta Stomach Content Analysis (Glahn et al. 1995)

Glahn et al. (1995) examined food habits of wintering cormorants in the Delta region of Mississippi. Birds were collected at catfish farms and night roosts. Stomach contents were analyzed, and total lengths of fish consumed were estimated. Channel catfish and gizzard shad were the only two important prey species in the diet, but proportions of these species in the diet varied dramatically among months, geographic locations, and between the sexes of birds collected. Overall, the diet was approximately equal between catfish and shad, though catfish was much more predominant in the diet of males, collected at both catfish farms and night roosts, than in the diet of females. The diet of birds collected at catfish farms did not vary between morning and afternoon collections, while samples from birds collected at roost sites were slightly lower in catfish biomass and more diverse in prey species. Thus, the authors assumed that stomach contents from birds collected at roost sites would form the best basis for assessing diet of wintering cormorant populations.

For the roost site samples, cormorants were collected from a total of 13 roost sites, six western and seven eastern. Spatial availability of catfish appeared to be an important factor determining the diet of cormorants among the roosting areas. The western roosts were located in five counties bordered by the Mississippi River and contained 6,835 ha of catfish ponds. The eastern roosts were located in five counties that contained 7,374 ha of catfish ponds. However, four of these roosts were within 5 km of Humphreys County, which has the highest catfish acreage in the U.S. In cormorants collected from western roosts, gizzard shad was the major prey item, while in those collected from eastern roosts, catfish was the major prey item. Glahn et al. (1995) assumed cormorants in the eastern roosts were consuming catfish in Humphreys County in addition to the other five counties, and because of this cormorants from the eastern roosts actually had access to a total catfish availability of 19,844 hectares. Compared with catfish availability at western roosts, this is an abundance ratio of 2.9:1. Additionally, differences in the diet of males and females may have influenced the difference in diet between regions, as males appeared to consume more catfish than females and to be more abundant at eastern roosts.

In both the farm and night roost collections, the size-class distributions of catfish recovered from cormorant stomachs fell within common commercial stocking sizes. The authors concluded that cormorants could have a negative impact on the commercial catfish industry in Mississippi, but that the proportion of catfish in the diet appeared to vary significantly with location and time of year (see Table 16, Table 17). The following strategies were suggested as potentially useful in reducing catfish losses: 1) increasing availability of shad by stocking these fish in areas of heavy catfish predation in the spring; 2) disperse roosting cormorants away from areas of high catfish density; 3) limit availability of size classes most frequently consumed (10 – 20 cm), particularly in the spring, possibly by increasing the size of “stocker” fish or delaying stocking until after cormorant migration in April.

Comments on study. While this study did not quantify the impact of cormorant predation, it did determine temporal and spatial magnitude of cormorant predation on catfish. It is also the first study to identify sex of the bird as an important source of variation in diet (D.V. Weseloh, pers. comm.). Such data are important in evaluating the appropriateness of management strategies to reduce potential impacts of cormorant predation.

Florida Foraging Behavior Observations (Schramm et al. 1984)

Schramm et al. (1984) conducted an observational study at a south Florida pond during September – October, 1980. The pond was stocked with 75,000 channel catfish fingerlings and observed by a person with binoculars between 0800 – 1200 and 1300 – 1700 on 20 randomly selected days. The mean catch per bird per hour and mean number of birds foraging during each hour were estimated. The sum of the mean total catch for the nine 1-hour periods (0800 – 1700) was used as a minimum estimate of average daily consumption. Schramm et al. (1984) reported 13 resident cormorants consumed an estimated 246 channel catfish daily during 0800 – 1700, which represents an average daily consumption of 19 catfish per bird, or an average of 304 g of catfish daily per bird. The 13 cormorants continued to feed at the pond during the winter and nested the following spring in a nearby cypress dome. By November of 1981, approximately 50 cormorants were feeding in the pond. Schramm et al. (1984) concluded that depredation by the Florida DCCO may be an important limiting factor to open-pond fish culture in Florida.

Comments on Study. Because Schramm et al. (1984) did not band or otherwise mark individuals, we are unsure of how the authors determined “residency” and breeding status of birds observed on the pond. Without marking the birds in some manner to identify individuals,

estimates of average consumption by resident birds may have a high degree of inaccuracy associated with them.

Controlled Catfish Predation Experiment (J. Glahn, unpubl. data)

J. Glahn (unpublished data from 1999 study) conducted a controlled experiment to assess DCCO impacts to gross production (in kg) of catfish. Ponds 0.1 acres in size were divided in half and stocked with catfish to mimic typical densities at southern catfish farms (i.e., 5,000 fingerlings / acre); an equal biomass of golden shiners was also stocked as a potential buffer prey (as a surrogate for shad). One DCCO was given access to one of the halves, while the other half received no predation. After a 10 day period of DCCO foraging, the birds were removed and catfish were cultured for the remainder of the 1999 growing season. In the ponds with DCCOs, greater biomass production per fish (compensatory growth) resulted; this was attributed to possibly greater relative food availability and density dependent growth. But despite compensatory growth in catfish, overall pond production was reported, on average, as 20 % less in treated versus control pond halves, suggesting that at the pond scale and predation rates investigated, DCCO impacts were additive. However, in a 0.1 acre pond where diseases depleted fish density, losses due to DCCO predation were insignificant. This study will be replicated in 2000 to supplement its statistical rigor.

Comments on Study. Preliminary results indicate that mortality due to DCCO predation can be additive under certain circumstances and compensatory in others. If these patterns are confirmed by data collected in 2000, researchers will be a large step closer to understanding net impacts of DCCOs on aquaculture. One important assumption in this study was that golden shiners would be preferentially selected by cormorants at a similar frequency as shad based on the similarity of handling time for these species; however no studies have been conducted to compare cormorant preference for these species. Because this is a controlled study at a much smaller scale and with fewer sources of mortality than are normally found in catfish ponds, more information will be needed to determine how or if these data should be used to estimate impacts at a larger scale. If it is true that 1) DCCO predation at ponds with diseased fish is compensatory and 2) DCCOs forage preferentially at ponds with diseased fish, then the influence of these factors will need to be addressed / incorporated in estimates of impact.

Bioenergetics Modeling

Mississippi Delta (Glahn and Brugger 1995)

Glahn and Brugger (1995) constructed a bioenergetics model to refine estimates of daily food requirements of wintering cormorants and their impact on the Mississippi Delta catfish industry. Their model estimated individual energy demands, population energy demands and catfish crop losses per month during two wintering periods: November to Apr 1989 – 1990 and November to Apr 1990 – 1991. The following parameters were incorporated in the model: basal metabolic rate (BMR, seasonally adjusted); daily activity budget; activity-specific metabolic rates; thermoregulation; daily energy budget; digestive efficiencies; individual energy demand; population estimates; population energy demand; nutrient composition of fish; diet composition; catfish mass consumed; percentage of catfish mass (by size class); catfish numbers consumed (by size class); and value (\$) of catfish (by size class). To transform the monthly energy demand of the cormorant population into the monthly populations of fish consumed, the following information was obtained: 1) the percent biomass of the diet of major fish groups consumed; 2)

the percent of the diet by size class and number of fish by weight in that size class; and 3) the energy (kJ) content in each fish group.

To assess economic and numerical impacts, two parameters from the Mississippi Delta catfish industry were included in the model: replacement value \$ loss of catfish, and “Standing crop” % loss. For the calculation of replacement cost of catfish consumed, the authors assumed \$20 US per thousand fish plus a variable cost of \$0.0049 to \$0.0069 / cm times the size (5 through 28 cm) of 12 commercial size classes in 2.54 cm increments. Larger fish (≥ 28 cm) were based on the number / kg times \$1.54 / kg. The standing crop of catfish in the size classes most often preyed upon by cormorants (10 to 23 cm) was difficult to identify; the number of fingerlings > 7 cm or exposed in ponds to cormorant predation was unknown. Therefore, to calculate an estimate of the total “standing crop” vulnerable to cormorant predation, the authors used the total area of catfish ponds stocked in the Mississippi Delta (c. 32,000 ha) times the average stocking rate of 12,350 fish / ha for all months except Mar and April. These latter months are the height of the stocking season, so the stocking rate of ponds was increased by 50 % or 18,525 catfish / ha for these months, resulting in “standing crops” of catfish vulnerable to predation of 395 million catfish from November to February and 592.5 million catfish in March and April.

Monthly cormorant populations varied (7,000 – 27,000 birds) and peaked in either February or March each year. Percent catfish in the diet also varied monthly (0 – 97 %), and diet showed a trend similar to populations with low catfish consumption in fall and peak catfish consumption February – April. Approximately 4 % (18 – 20 million catfish) of the estimated catfish standing crop was consumed by cormorants each year, and annual losses to the catfish industry were predicted at \$2 million and \$1.8 million during 1989 – 1990 and 1990 – 1991, respectively. More than half of the average annual consumption of catfish by cormorants was predicted to occur in February and March due to increasing cormorant populations and a shift in the diet towards more catfish during those months. Glahn and Brugger (1995) note that their average annual monetary loss figure of \$1.9 million was 40 % less than the annual loss of \$3.3 million estimated by Stickleby and Andrews (1989; see above), and that their model would also have produced much higher losses if they had assumed a constant high population of cormorants subsisting on a diet of 100 % catfish.

Comments on Study. Glahn and Brugger (1995) believe that their model provides the best estimate of the impact of DCCO predation on the Mississippi Delta catfish industry during the winters of 1989 – 1990 and 1990 – 1991, based on the empirical data assembled to construct the model. We agree with this statement. Of all the literature we reviewed that attempted to assess impacts to fish populations (including sport and commercial fish populations), this approach appeared to be one of the most thorough and inclusive. The authors incorporated several important physiological and ecological parameters into the model, data necessary to more accurately represent energy flow between predator and prey.

Nevertheless, both estimates of impact (numerical and economic) are based on parameters which make it difficult to assess the impacts precisely. The standing crop estimate was based in part on an unpublished average stocking rate, which we were unable to judge for accuracy. [Estimates of fish stocking densities range widely; Hodges (1989) reported a range of fish density, from 5,000 – 150,000 fish / ha reported by farmers.] Economic impacts were reported in terms of replacement costs, which, as described above, can not be extrapolated to net impacts at time of harvest, because the impact of cormorants relative to other mortality factors has not been determined.

Conclusions: Impacts to Aquaculture

Studies to assess impacts of DCCOs on aquaculture have evaluated many diverse techniques; all suggest that DCCOs can consume large amounts of catfish at certain times and locations. However, the magnitude of these impacts still needs to be made more clear. Similar questions that remain unanswered regarding impacts to sport and commercial fisheries are just beginning to be addressed in studies assessing impacts to aquaculture [e.g., compensatory vs. additive mortality, influence of other mortality factors, compensatory mechanisms in farm fish populations; see *Controlled Catfish Predation Experiment (J. Glahn, unpubl. data)*, above]. A more comprehensive understanding of the interrelationships between fish-eating birds and catfish farming is needed. More studies of the interactions within this system are necessary to understand the net effect of cormorants at catfish ponds, and the nature and magnitude of cormorant predation. Jackson and Jackson (1995) suggested that one way to gain a more comprehensive understanding of the impact of cormorants on ponds is to randomly sample all available ponds, not just those at which cormorants are seen feeding. This is important in trying to assess impacts because: 1) there may be characteristics of ponds that attract or discourage cormorants (Hodges 1989); 2) by feeding at ponds where fish are weakened or dying (Hodges 1989), cormorants may actually benefit some farmers (Jackson and Jackson 1995); and 3) estimates of losses to cormorants may be inflated as a result of inclusion of losses that would have occurred with or without cormorants present.

The bioenergetics modeling and DCCO inclusion / exclusion experiments were two of the more promising techniques utilized. If modeling efforts could be expanded to incorporate parameters such as fish growth rates in the presence / absence of DCCOs, oxygen stress, other predators, and disease, a more comprehensive view of the system and how factors interact might be possible, and provide more relevant information on the net effect of cormorants at ponds. Such work may also provide valuable insights for impact assessment of open-water fisheries.

Studies on food preferences

In their review of cormorant diet literature, Trapp et al. (1995) found that most studies reveal little about food preferences; this requires simultaneous information on the relative abundance of all potential prey species at a given site. The only study we found that examined food preferences by utilizing quantitative data on prey fish species was the one described above by Glahn et al. (1998). At both lakes cormorants fed mainly on shad and sunfishes. For the most part, cormorants appeared to consume the fish species most available; but at Lake Beulah, cormorants did appear to have a preference for sunfishes, particularly bluegill. But because of possible biases in fish sampling techniques (see above) the preference may have been overstated. The authors also found that cormorants appeared to prefer gizzard shad over threadfin shad, and they speculated that this is because gizzard shad represent a more optimally-sized prey for cormorants. At Lake Eufala, where threadfin shad greatly outnumbered gizzard shad, threadfin shad were taken proportional to their relative abundance. Glahn et al. (1998) concluded that availability, rather than size, is probably the most important factor in prey selection by cormorants; another factor may be accessibility of prey fish.

Stickley et al. (1992) reported a preference for gizzard shad over catfish. In the only pond that contained both gizzard shad and catfish, observers noted 64 shad and only two catfish in cormorant bills over about three hours. Hodges (1989) also reported a possible preference for shad over catfish.

Table 7. Pacific Coast: Occurrence of Salmonids in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

Location	% in diet	Time	Assessment Method	Source	Variation
<i>Oncorhynchus spp.</i>					
Lower Columbia River: Estuary colonies combined	30.3 (N); 23.7 (B)	Apr-Aug	Boli ($n = 64$); Stomach contents ($n = 90$)	Roby et al. 1998	Temporal
	14.4 (B)	Apr	Boli, Stomach contents ($n = 19$ combined)		
	95.3 (B)	May	Boli, Stomach contents ($n = 19$ combined)		
	6.6 (B)	June	Boli, Stomach contents ($n = 62$ combined)		
	4.3 (B)	July	Boli, Stomach contents ($n = 54$ combined)		
L. Columbia River: Upriver	35 (B) ¹	Apr-Aug	Boli, Stomach contents ($n = 120$ combined)	Roby et al. 1998	Spatial
L. Columbia River: River mouth	16 (B) ¹	Apr-Aug	Boli, Stomach contents ($n = 31$ combined)		Spatial
British Columbia, Mandarte Is.	0.2 (N); 0.9 (B)	Nesting	Boli (n not provided)	Robertson 1974	

¹ Diet composition from East Sand Island, near the Columbia River mouth, was based on a small sample size; most stomach content samples were collected farther up river in the Rice Island area, thus the diet composition data for cormorants is probably biased toward salmonids (Roby et al. 1998).

Table 8. Pacific Coast: Occurrence of littoral or estuarine fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

Location	% in diet	Time	Assessment Method	Source	Variation
STICKLEBACK					
Lower Columbia River, estuary colonies	21.2 (N); 1.6 (B)	Apr-Aug	Boli (<i>n</i> = 64); Stomach contents (<i>n</i> = 90)	Roby et al. 1998	
Lower Columbia River: Upriver	24 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 120 combined)		Spatial
L. Columbia River: River mouth	14 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 31 combined)		
British Columbia, Mandarte Is.	0.4 (N); 0.1 (B)	Nesting	Boli (<i>n</i> not provided)	Robertson 1974	
SHINER PERCH¹					
L. Columbia River, estuary colonies	8.6 (N); 3.5 (B)	Apr-Aug	Boli (<i>n</i> = 64); Stomach contents (<i>n</i> = 90)	Roby et al. 1998	
Lower Columbia River: Upriver	4 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 120 combined)		Spatial
L. Columbia River: River mouth	22 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 31 combined)		
British Columbia, Mandarte Is.	15.5 (N); 20.5 (B)	Nesting	Boli (<i>n</i> not provided)	Robertson 1974	
California, Central Coast, Farallon Is.	78.6 (N)	Post-breeding season	Pellets (<i>n</i> = 175)	Ainley et al. 1981	Spatial, Assessment method
California, San Miguel I.	12.3 (N)	Breeding season	Boli (<i>n</i> = 39)		
Baja California, I. San Martin	7.1 (N)	Post-breeding season	Pellets (<i>n</i> = 35)		

Location	% in diet	Time	Assessment Method	Source	Variation
STRIPED SEAPERCH					
British Columbia, Mandarte Is.	1.3 (N); 3.5 (B)	Nesting	Boli (<i>n</i> not provided)	Robertson 1974	
California, Central Coast, Farallon Is.	0.1 (N)	Post-breeding season	Pellets (<i>n</i> = 175)	Ainley et al. 1981	Spatial, Assessment method
Other Embiotocids (not including SHINER PERCH and STRIPED PERCH, above)					
California, Central Coast, Farallon I.	13.8 (N)	Post-breeding season	Pellets (<i>n</i> = 175)	Ainley et al. 1981	Spatial, Assessment method
California, San Miguel I.	8.9 (N)	Breeding season	Boli (<i>n</i> = 39)		
Baja California, I. San Martin	2.2 (N)	Post-breeding season	Pellets (<i>n</i> = 35)		
PACIFIC SAND LANCE¹					
British Columbia, Mandarte Is.	20.5 (N); 4.6 (B)	Nesting	Boli (<i>n</i> not provided)	Robertson 1974	
PEAMOUTH					
L. Columbia River, estuary colonies	15.8 (N); 32.9 (B)	Apr-Aug	Boli (<i>n</i> = 64); Stomach contents (<i>n</i> = 90)	Roby et al. 1998	Spatial
Lower Columbia River: Upriver	19 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 120 combined)		
Lower Columbia River: River mouth	8 (B)	Apr-Aug	Boli, Stomach contents (<i>n</i> = 31 combined)		

1 During summer, shiner perch and Pacific sand lance inhabit shallow water in large schools, and are probably caught while in this zone (Robertson 1974).

Table 9. Pacific Coast: Occurrence of littoral-benthic fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

Location	% in diet	Time	Assessment Method	Source	Variation
Cottidae (SCULPIN SP.)					
L. Columbia River, estuary colonies	13.5 (N); 6.0 (B)	Apr-Aug	Boli (n = 64); Stomach contents (n = 90)	Roby et al. 1998	Spatial
L. Columbia River: Upriver	8 (B)	Apr-Aug	Boli, Stomach contents (n = 120 combined)		
L. Columbia River: River mouth	30 (B)	Apr-Aug	Boli, Stomach contents (n = 31 combined)		
British Columbia, Mandarte Is.	2.7 (N); 5.9 (B)	Nesting	Boli (n not provided)	Robertson 1974	
California, Central Coast, Farallon Is.	3 (N)	Post-breeding season	Pellets (n = 175)	Ainley et al. 1981	
PENPOINT GUNNEL					
British Columbia, Mandarte Is.	23.8 (N); 35.7 (B)	Nesting	Boli (n not provided)	Robertson 1974	
CRESCENT GUNNEL					
British Columbia, Mandarte Is.	22.8 (N); 15.9 (B)	Nesting	Boli (n not provided)	Robertson 1974	
SNAKE PRICKLEBACK					
British Columbia, Mandarte Is.	11.5 (N); 10.2 (B)	Nesting	Boli (n not provided)	Robertson 1974	
SUCKER					
L. Columbia River, estuary colonies	2.3 (N); 8.1 (B)	Apr-Aug	Boli (n = 64); Stomach contents (n = 90)	Roby et al. 1998	

Table 10. Pacific Coast: Occurrence of pelagic fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

Location	% in diet	Time	Assessment Method	Source	Variation
PACIFIC HERRING					
British Columbia, Mandarte Is.	1.3 (N); 2.7 (B)	Nesting	Boli (<i>n</i> not provided)	Robertson 1974	
Lower Columbia River, estuary colonies	0.2 (N) 0.5 (B)	Apr-Aug	Boli (<i>n</i> = 64); Stomach contents (<i>n</i> = 90)	Roby et al. 1998	
CLUPEIDS (<i>Opisthonema spp.</i>)					
Gulf of California, Bahia de los Angeles	0.2 (N)	Post-breeding	Pellets (<i>n</i> = 106)	Ainley et al. 1981	Spatial
Sonora, Bahia de Kino	11.6 (N)	Post-breeding	Pellets (<i>n</i> = 208)		

Table 11. Great Lakes: Occurrence of open-fresh water fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
ALEWIFE					
L. Huron, Michigan & Superior ¹	41 (N); 57 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	Spatial
	100 (F)	Aug			
Green Bay, n. Lake Michigan	76 total fish biomass consumed)	May-Aug			
Beaver Is., n. Lake Michigan	72 (total fish biomass consumed)				
Main body, n. Lake Huron	64 (total fish biomass consumed)				
N. Channel / Georgian Bay, L. Huron	37 (total fish biomass consumed)				
L. Huron and Ontario	0-50 (F)	Apr/May-July/Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial, Diet Assessment
L. Ontario, Little Galloo	47 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	13 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Lake Huron, Bustard Is.	28 (F)	June 20, 1994	Pellets (<i>n</i> = 36)		
	70 (F)		Boli (<i>n</i> = 47)		
Little Galloo Is., Lake Ontario	27 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	1 (N)	May	Pellets (<i>n</i> = 742)		
	68 (N)	June	Pellets (<i>n</i> = 601)		
Les Chenaux Is., n. Lake Huron	29 (N); 47 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	< 0.1 (N, B)	Apr 17-May 16	Stomach contents (<i>n</i> not reported)		
	61 (N); 90 (B)	Jul 4-Aug 1	Stomach contents (<i>n</i> not reported)		
w. Lake Erie	0.4 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
Cat Is., Lake Michigan	52.5 ² (F)	Jul 5, 7, 23; Aug 2, 5, 12, 1993	Boli (<i>n</i> = 507)	Matteson et al. 1999 ³	Temporal, Spatial
	57.5 (F)	Jun 3, 26; Jul 10, 25, 1997	Boli (<i>n</i> = 400)		
Willow Is., L. Michigan	50.8 (F)	Jun 13-July 25, 1983	Boli (<i>n</i> = 1090)		
GIZZARD SHAD					
W. Lake Erie	70 (N.); 50 (F); 48 (B)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	
RAINBOW SMELT					
L. Huron, Michigan and Superior	7 (N); 8 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
Apostle Is., L. Superior (Gull Island)	1 (F)	May-Sept / Oct 1984	Pellets, Boli (<i>n</i> = 150)	Craven and Lev 1987	
Apostle Is., L. Superior (Eagle I.)	0.9 (F)	May-Sept / Oct 1984	Pellets (<i>n</i> not reported)		
Apostle Is., (Gull Island)	1.8 (F)	May-Sept / Oct 1985	Boli (<i>n</i> not reported)	Matteson et al. 1999	
L. Huron and Ontario	2 - 65 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial, Temporal
L. Ontario, Little Galloo	4 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	57 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Les Chenaux Islands, n. Lake Huron	0.7 (N); 0.2 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	
	< 0.1 (N, B)	Aug 2- 19	Stomach contents (<i>n</i> not reported)		
	2 (N); 0.4 (B)	Jul 4-Aug 1	Stomach contents (<i>n</i> not reported)		
w. Lake Erie	1 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	

1 Food samples collected at L. Superior colonies were too few to be useful. However, of 21 food items recovered at a L. Superior colony, only 10 % (*n* = 2) were alewives. In conjunction with Craven and Lev's (1987) study, this suggests that in L. Superior alewife is much less prominent in the cormorant diet than in Lakes Huron and Michigan. This is also true for w. Lake Erie (Bur et al. 1999).

2 Sample represents both alewife and gizzard shad combined (Matteson et al. 1999).

3 This study reports frequency of occurrence of fish in DCCO regurgitation samples from 5 Wisconsin studies on L. Superior and Michigan conducted intermittently during 1983 – 1997.

Table 12. Great Lakes: Occurrence of littoral-freshwater fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
STICKLEBACKS (NINESPINE, THREESPINE)					
L. Huron, Michigan and Superior	41 (N); 3 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
Apostle Is., L. Superior (Gull Island)	20.2 (F)	May-Sept / Oct 1984	Pellets, Boli (<i>n</i> not reported)	Craven and Lev 1987	Spatial; Temporal; Diet assessment method?
Apostle Is., L. Superior (Eagle I.)	NA ¹	May-Sept / Oct 1984	Pellets (<i>n</i> not reported)		
Apostle Is., L. Superior (Gull Island)	75.9 (F)	May-Sept / Oct 1985	Boli (<i>n</i> not reported)	Matteson et al. 1999	
Little Galloo Is., Lake Ontario	22 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	51 (N)	May	Pellets (<i>n</i> = 742)		
	0 (N)	Sept	Pellets (<i>n</i> = 301)		
Les Chenaux Is., n. Lake Huron	58 (N); 15 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	91 (N); 46 (B)	May 17-Jul 3	Stomach contents (<i>n</i> not reported)		
	0.7 (N); 0.2 (B)	Sep 14-Oct 6	Stomach contents (<i>n</i> not reported)		
SMALLMOUTH BASS					
L. Huron, Michigan and Superior	< 1 (N); 1 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
L. Huron and Ontario	2 - 88 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	6 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	41 (F)	May-Aug	Pellets (<i>n</i> = 144)		
w. Lake Erie	0.8 (F); 0.3 (B)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	
Cat Is., Lake Michigan	0.2 (F)	Jun 3, 26; Jul 10, 25, 1997	Boli (<i>n</i> = 400)	Matteson et al. 1999	

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
Little Galloo Is., Lake Ontario	1.5 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	0.3 (N)	April	Pellets (<i>n</i> = 276)		
	3 (N)	July, Aug	Pellets (<i>n</i> = 1316)		
ROCK BASS					
L. Huron, Michigan and Superior	< 1 (N.); 4 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
L. Huron and Ontario	0.8 - 14 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	1.1 (F)	Apr-July	Pellets (<i>n</i> = 938)		
L. Huron, Bustard Is.	9.7 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Little Galloo Is., Lake Ontario	3 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	3.6 (N)	May	Pellets (<i>n</i> = 742)		
	2.1 (N)	June	Pellets (<i>n</i> = 601)		
PUMPKINSEED					
L. Ontario, Little Galloo	9.6 (F)	Apr-July	Pellets (<i>n</i> = 938)	Neuman et al. 1997	
Little Galloo Is., Lake Ontario	6.8 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	18.1 (N)	April	Pellets (<i>n</i> = 276)		
	3.1 (N)	July	Pellets (<i>n</i> = 715)		
W. Lake Erie	0.8 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	
Centrarchidae (SMALLMOUTH BASS, ROCK BASS, PUMPKINSEED)					
Les Chenaux Is., n. Lake Huron	0.8 (N); 6 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	9 (N); 15 (B)	Apr 17-May 16	Stomach contents (<i>n</i> not reported)		
	0.1 (N); 0.2 (B)	Jul 4-Aug 1	Stomach contents (<i>n</i> not reported)		
W. Lake Erie ²	0.8 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	
YELLOW PERCH					
L. Huron, Michigan and Superior	3 (N); 13 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
L. Huron and Ontario	4 - 78 (F)	May-Aug	Pellets ($n = 1595$)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	20 (F)	Apr-July	Pellets ($n = 938$)		
Lake Huron, Bustard Is.	17 (F)	May-Aug	Pellets ($n = 144$)		
Les Chenaux Is., n. Lake Huron	2 (N); 11 (B)	Apr-Oct	Stomach contents ($n = 373$)	Belyea et al. 1999	Temporal
	< 0.1 (N, B)	Aug 20-Sep 13	Stomach contents (n not reported)		
	42 (N); 48 (B)	Apr 17-May 16	Stomach contents (n not reported)		
Little Galloo Is., Lake Ontario	18 (N)	Apr-Sept	Pellets ($n = 2939$)	Johnson et al. 1999	Temporal
	7 (N)	June	Pellets ($n = 601$)		
	48 (N)	Sept	Pellets ($n = 301$)		
W. Lake Erie	0.2 (N); 1 (B); 5 (F)	Apr-Oct	Stomach contents ($n = 302$)	Bur et al. 1999	
Cat Is., Lake Michigan	46.1 (F)	Jul 5, 7, 23; Aug 2, 5, 12, 1993	Boli ($n = 507$)	Matteson et al. 1999	Temporal, Spatial
	13.7 (F)	Jun 3, 26; Jul 10, 25, 1997	Boli ($n = 400$)		
Willow Is., L. Michigan	38.7 (F)	Jun 13-July 25, 1983	Boli ($n = 1090$)		
EMERALD SHINER					
W. Lake Erie	27 (N); 9 (B); 34 (F)	Apr-Oct	Stomach contents ($n = 302$)	Bur et al. 1999	
COMMON SHINER					
L. Huron, Michigan and Superior	2 (N); < 1 (B)	May-Aug	Boli (n not reported)	Ludwig et al. 1989	
SPOTTAIL SHINER					
L. Ontario, Little Galloo	0.9 (F)	Apr-July	Pellets ($n = 938$)	Neuman et al. 1997	
Cat Is., ³ L. Michigan	16.2 (F)	Jun 3, 26; Jul 10, 25, 1997	Boli ($n = 400$)	Matteson et al. 1999	Temporal
Willow Is., ³ L. Michigan	5.1 (F)	Jun 13-July 25, 1983	Boli ($n = 1090$)		

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
<i>Notropis sp. (SHINER SP.)</i>					
L. Huron and Ontario	8 – 65 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	9 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	58 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Cyprinidae					
Little Galloo Is., Lake Ontario	13 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	5 (N)	June	Pellets (<i>n</i> = 601)		
	32 (N)	Sept	Pellets (<i>n</i> = 301)		
Les Chenaux Is., n. Lake Huron	2 (N); 2 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	24 (N); 4 (B)	Apr 17-May 16	Stomach contents (<i>n</i> not reported)		
	< 0.1 (N, B)	May 17-Jul 3	Stomach contents (<i>n</i> not reported)		
WHITE PERCH					
L. Huron and Ontario	7 - 27 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	18 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	0 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Cat Is., Lake Michigan	0.2 (F)	Jul 5, 7, 23; Aug 2, 5, 12, 1993	Boli (<i>n</i> = 507)	Matteson et al. 1999	Temporal
	10 (F)	Jun 3, 26; Jul 10, 25, 1997	Boli (<i>n</i> = 400)		
W. Lake Erie	0.1 (N); 0.2 (B); 2.0 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	

1 Only pellets were collected on Eagle Is., and ninespine sticklebacks could only be identified in boli. However, ninespine stickleback was abundant around this site and probably consumed by DCCOs there (Craven and Lev 1987).

2 Unidentified Centrarchid species (Bur et al. 1999).

3 Cat and Willow Islands are about 400 m apart (Matteson et al. 1999).

Table 13. Great Lakes: Occurrence of bottom dwelling-fresh water fish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
SCULPIN (SLIMY, SPOONHEAD)					
L. Huron, Michigan and Superior	4 (N); 1 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
L. Huron and Ontario	3-41 (F)	Apr/May-July/Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial, Diet Assessment
L. Ontario, Little Galloo	5 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	40 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Lake Huron, Bustard Is.	44 (F) 6 (F)	June 20, 1994	Pellets (<i>n</i> = 36) Boli (<i>n</i> = 47)		
Little Galloo Is., Lake Ontario	4 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	8 (N)	April	Pellets (<i>n</i> = 742)		
	1 (N)	Sept	Pellets (<i>n</i> = 301)		
Les Chenaux Is., n. Lake Huron	5 (N); 4 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	< 0.1 (N,B)	Sept 14-Oct 6	Stomach contents (<i>n</i> not reported)		
	42 (N); 24 (B)	Aug 2-19	Stomach contents (<i>n</i> not reported)		
Apostle Is., L. Superior (Gull Island)	21.8 (F)	May-Sept / Oct 1984	Pellets, Boli (<i>n</i> = 150)	Craven and Lev 1987	Spatial; Temporal; Diet assessment method
Apostle Is., L. Superior (Eagle Is.)	58 (F)	May-Sept / Oct 1984	Pellets (<i>n</i> not reported)		
Apostle Is., L. Superior (Gull Island)	1.8 (F)	May-Sept / Oct 1985	Boli (<i>n</i> not reported)	Matteson et al. 1999	

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
TROUT-PERCH					
Lakes Huron and Ontario	3 - 41 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
Lake Ontario, Little Galloo	16 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	28 (F)	May-Aug	Pellets (<i>n</i> =144)		
Little Galloo Is., Lake Ontario	2 (N)	Apr-Sept	Pellets (<i>n</i> = 2939)	Johnson et al. 1999	Temporal
	6 (N)	April	Pellets (<i>n</i> = 276)		
	0 (N)	Sept	Pellets (<i>n</i> = 301)		
Apostle Is., L. Superior (Gull Island)	9.5 (F)	May-Sept / Oct	Pellets, Boli (<i>n</i> = 150)	Craven and Lev 1987	Spatial, Diet assessment ?
Apostle Is., L. Superior (Eagle I.)	1.8 (F)	May-Sept / Oct	Pellets (<i>n</i> not reported)		
WHITE SUCKER					
L. Huron, Michigan and Superior	< 1 (N); 7 (B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
L. Huron and Ontario	1 - 49 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	1 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	36 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Les Chenaux Is., n. Lake Huron	0.2 (N); 4.8 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	Temporal
	2 (N); 12 (B)	Apr 17-May 16	Stomach contents (<i>n</i> not reported)		
	< 0.1 (N,B)	May 17-Jul 3	Stomach contents (<i>n</i> not reported)		

Table 14. Great Lakes: Occurrence of Salmonids in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
COREGONUS SP. (LAKE WHITEFISH, LAKE HERRING)					
L. Huron, Michigan and Superior	< 1 (N, B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
L. Huron and Ontario	0.3 - 22 (F)	May-Aug	Pellets (<i>n</i> = 1595)	Neuman et al. 1997	Spatial
L. Ontario, Little Galloo	0.3 (F)	Apr-July	Pellets (<i>n</i> = 938)		
Lake Huron, Bustard Is.	0.7 (F)	May-Aug	Pellets (<i>n</i> = 144)		
Apostle Is., L. Superior (Gull Island)	1.6 (F)	May-Sept / Oct	Pellets, Boli (<i>n</i> not reported)	Craven and Lev 1987	
Apostle Is., L. Superior (Eagle I.)	2.7 (F)	May-Sept / Oct	Pellets (<i>n</i> not reported)		
LAKE TROUT					
L. Huron, Michigan and Superior	< 1 (N,B)	May-Aug	Boli (<i>n</i> not reported)	Ludwig et al. 1989	
Apostle Is., L. Superior (Gull Island)	1 (F)	May-Sept / Oct	Pellets, Boli (<i>n</i> not reported)	Craven and Lev 1987	
Apostle Is., L. Superior (Eagle I.)	2 (F)	May-Sept / Oct	Pellets (<i>n</i> not reported)		
L. Ontario, E. Basin ¹	0.2 (N)	Apr-mid July	Pellets (<i>n</i> =4,848)	Ross and Johnson 1999a	
Salmonidae (BLOATER, LAKE WHITEFISH, ROUND WHITEFISH, SPLAKE)					
Les Chenaux Is., n. Lake Huron	0.1 (N); 0.9 (B)	Apr-Oct	Stomach contents (<i>n</i> = 373)	Belyea et al. 1999	
	< 0.1 (N,B)	Apr 17- May 16	Stomach contents (<i>n</i> not reported)		
	0.2 (N); 1.4 (B)	Jul 4- Aug 1	Stomach contents (<i>n</i> not reported)		

¹ data provided on % composition of salmonines in diet, but almost all identified were stocked lake trout.

Table 15. Great Lakes: Locally important fish species in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Lake(s) / Island	% in diet	Time	Assessment Method	Source	Variation
BURBOT					
Apostle Is., L. Superior (Gull Island)	15.5 (F)	May-Sept / Oct, 1984	Pellets, Boli (<i>n</i> = 150)	Craven and Lev 1987	Spatial; Temporal; Diet assessment method
Apostle Is., L. Superior (Eagle I.)	4 (F)	May-Sept / Oct, 1984	Pellets (<i>n</i> not reported)		
Apostle Is., Lake Superior (Gull island)	9.3 (F)	May-Sept / Oct, 1985	Boli (<i>n</i> not reported)	Matteson et al. 1999	
FRESHWATER DRUM					
W. Lake Erie	1 (N); 33 (B); 34 (F)	Apr-Oct	Stomach contents (<i>n</i> = 302)	Bur et al. 1999	
LAKE / NORTHERN CHUB					
Apostle Is., L. Superior (Gull Island)	11 (F)	May-Sept / Oct 1984	Pellets, Boli (<i>n</i> = 150)	Craven and Lev 1987	Spatial; Temporal; Diet assessment method
Apostle Is., L. Superior (Eagle I.)	6 (F)	May-Sept / Oct	Pellets (<i>n</i> not reported)		
Apostle Is., L. Superior (Gull Island)	2.8 (F)	May-Sept / Oct 1985	Boli (<i>n</i> not reported)	Matteson et al. 1999	

Table 16. Southern U.S.: Occurrence of Shad in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

Location	% in diet	Time	Assessment Method	Source	Variation
GIZZARD SHAD					
TX, inland reservoirs	9.9 (N); 14.3 (B)	Nov 1986 – Mar 1987	Stomach contents (n = 420)	Campo et al 1993	
THREADFIN SHAD					
TX, inland reservoirs	69.3 (N); 11.8 (B)	Nov 1986 – Mar 1987	Stomach contents (n = 420)	Campo et al 1993	
SHAD SP. (GIZZARD, THREADFIN)					
Lake Beulah, MS	89.4 (F); 93.4 (N)	Dec 1996 – Mar 1997	Stomach contents (n = 123)	Glahn et al. 1998	Spatial
Lake Eufala, AL	70.6 (F); 77.8 (N)	Mar-Apr 1997	Stomach contents (n = 51)		
MS Delta Region, Catfish Farms	100 (F, B)	Oct 1987, 1988	Stomach contents (n = 5)	Glahn et al. 1995	Temporal, Spatial
	0 (F, B)	Jan 1988, 1989	Stomach contents (n = 12)		
	2 (F); 0.6 (B)	Mar 1988, 1989	Stomach contents (n = 49)		
	9.1 (F); 5.3 (B)	Oct – Mar 1987/88 Oct – Mar 1988/89	Stomach contents (n = 66)		
MS Delta Region, Night Roosts	40.2 (F); 40.8 (B)	Winter 1989-90	Stomach contents (n = 204)		
	47.5 (F); 44 (B)	Winter 1990-91	Stomach contents (n = 257)		
Central and se. AR, aquaculture facilities	71.6 (F)	Oct 18-Dec 5 1988	Stomach contents (n = 135)	Bivings et al. 1989	

Table 17. Southern U.S.: Occurrence of Catfish in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

V = percent volume is equivalent to % biomass, except that quantities are measured volumetrically

Location	% in diet	Time	Assessment Method	Source	Variation
CATFISH					
TX, inland reservoirs	1.1 (N); 9.6 (B)	Nov 1986 – Mar 1987	Stomach contents (n = 420)	Campo et al 1993	
Lake Beulah, MS	8.9 (F); 1.8 (N)	Dec 1996 – Mar 1997	Stomach contents (n = 123)	Glahn et al. 1998	Spatial
Lake Eufala, AL	29.4 (F); 11.4 (N)	Mar-Apr 1997	Stomach contents (n = 51)		
MS Delta Region, Catfish Farms	0 (F, B)	Oct 1987, 88	Stomach contents (n = 5)	Glahn et al. 1995	Temporal, Spatial
	50 (F); 48.6 (B)	Jan 1988, 89	Stomach contents (n = 12)		
	91 (F); 95 (B) ¹	Mar 1988, 89	Stomach contents (n = 49)		
	77.3 (F); 85.3 (B) ¹	Oct – Mar 1987/88 Oct – Mar 1988/89	Stomach contents (n = 66)		
MS Delta Region, Night Roosts	55.4 (F); 50.7 (B)	Winter 1989/90	Stomach contents (n = 204)		
	54.8 (F); 50.1 (B)	Winter 1990/91	Stomach contents (n = 257)		
Central and se. AR, aquaculture facilities	6.8 (F)	Oct 18 - Dec 5 1988	Stomach contents (n = 135)	Bivings et al. 1989	

¹ Because 74 % of the samples was obtained in March, and the diet from March collections was almost all catfish, the disproportional sampling appeared to skew the overall diet towards catfish.

Table 18. Southern U.S.: Occurrence of Centrarchids in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample
 B = biomass of taxon of interest as a percent of total sample biomass
 F = percent of samples which contain at least one specimen of the taxon of interest

Location	% in diet	Time	Assessment Method	Source	Variation
SUNFISH					
TX, inland reservoirs	7.8 (N); 15 (B) ¹	Nov 1986 – Mar 1987	Stomach contents (n = 420)	Campo et al. 1993	
Cent. and se. AR, aqua. facilities	12.9 (F) ¹	Oct 18 – Dec 5 1988	Stomach contents (n = 135)	Bivings et al. 1989	
Lake Beulah, MS	17.3 (F); 3.3 (N)	Dec 1996 – Mar 1997	Stomach contents (n = 123)	Glahn et al. 1998	Spatial
Lake Eufala, AL	17.6 (F); 3.7 (N)	Mar-Apr 1997	Stomach contents (n = 51)		
Centrarchidae					
Lake Eufala, AL	31.7 (F); 6.8 (N)	Mar-Apr 1997	Stomach contents (n = 51)	Glahn et al. 1998	
BREAM²					
MS Delta Region, Catfish Farms	0 (F,B)	Oct 1987-88	Stomach contents (n = 5)	Glahn et al. 1995	Temporal, Spatial
	50 (F); 51.4 (B)	Jan 1988-89	Stomach contents (n = 12)		
	8 (F); 3.5 (B)	Mar 1988-89	Stomach contents (n = 49)		
	15.2 (F); 9.4 (B)	Oct – Mar 1987/88 Oct – Mar 1988/89	Stomach contents (n = 66)		
MS Delta Region, Night Roosts	12.3 (F); 5.2 (B)	Winter 1989-90	Stomach contents (n = 204)		
	11.7 (F); 4.5 (B)	Winter 1990-91	Stomach contents (n = 257)		

1 ≥ 47 % bluegill.

2 Includes sunfish and crappie species.

Table 19. Southern U.S.: Locally important fish species in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample
 B = biomass of taxon of interest as a percent of total sample biomass

Location	% in diet	Time	Assessment Method	Source	Variation
BLUE TILAPIA					
TX, inland reservoirs	5.2 (N); 18.2 (B)	Nov 1986 – Mar 1987	Stomach contents (n = 420)	Campo et al. 1993	

Table 20. Atlantic Region: Occurrence of Marine bottom fish in the diet of Double-crested Cormorants.

B = biomass of taxon of interest as a percent of total sample biomass
 F= percent of samples which contain at least one specimen of the taxon of interest
 N = number of specimens of the taxon of interest as a percent of all specimens in sample
 R=relative importance, sum of % of each prey species divided by number of stomachs analyzed

Location	% in diet	Time	Assessment Method	Source	Variation
COD SP. (92% ATLANTIC COD)					
Eastern Shore rivers, Nova Scotia	37.7 (F); 43.1 (N)	Breeding season, 1979-80	Pellets, Boli (n not reported)	Milton et al. 1995	
SCULPIN SP.					
Eastern Shore rivers, Nova Scotia	33.6 (F); 17.5 (N)	Breeding season, 1979-80	Pellets, Boli (n not reported)	Milton et al. 1995	
Penobscot River, ME, below head of tide	3.4 (F)	Late Apr 1986-88	(n = 66)	Blackwell et al. 1997 ¹	
	0 (F)	May 1986-88	(n = 244)		
	5.3 (F)	Early June 1986	(n = 19)		
	0 (F)	Late Apr 1993	(n = 7)		
	2.6 (F); 1.5 (N)	May 1992-93	(n = 68)		
	0.85 (F); 0.5 (N)	Jun 1992-93	(n = 73)		
Passamaquoddy Bay, New Brunswick	53.5 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	

Location	% in diet	Time	Assessment Method	Source	Variation
Estuary of St. Lawrence River, Québec	8 (F); 2.6 (N)	Jun 4-Jul 14, 1994	Boli (<i>n</i> = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	4 (F); 1.5 (N)	Jun 4-Jul 14, 1995	Boli (<i>n</i> = 301)		
Gulf of St. Lawrence River, St. Mary Is., Québec	4.3 (F); 1 (N)	Jun 28-Jul 20, 1995	Boli (<i>n</i> = 69)		
	2.5 (F); 0.5 (N)	Jun 28-Jul 20, 1996	Boli (<i>n</i> = 118)		
Bays of Prince Edward Island	11.9 (B)	Breeding season, 1985	Stomach contents (<i>n</i> not reported)	Birt et al. (1987)	
GUNNEL SP.					
Estuary of St. Lawrence River, Québec	40 (F); 19.9 (N)	Jun 4-Jul 14, 1994	Boli (<i>n</i> = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	13.6 (F); 6.7 (N)	Jun 4-Jul 14, 1995	Boli (<i>n</i> = 301)		
Gulf of St. Lawrence River, St. Mary Is., Québec	10.1 (F); 3.7 (N)	Jun 28-Jul 20, 1995	Boli (<i>n</i> = 69)		
	10.2 (F); 3.6 (N)	Jun 28-Jul 20, 1996	Boli (<i>n</i> = 118)		
E. Shore rivers, Nova Scotia	12.3 (F); 16.3 (N)	Breeding season, 1979-1980	Pellets, Boli (<i>n</i> not reported)	Milton et al. 1995	
Passamaquody Bay, New Brunswick	4.9 (R)	May 19-31 1986	Stomach contents (<i>n</i> = 117)	Kehoe 1987 (<i>in</i> Cairns 1998)	
Penobscot R., ME, below head of tide	1.7 (F)	Early Jun 1992 – 1993	(<i>n</i> = 58)	Blackwell et al. 1997	
CUNNER					
Eastern Shore rivers, Nova Scotia	12.3 (F); 4.1 (N)	Breeding season, 1979-1980	Pellets, Boli (<i>n</i> not reported)	Milton et al. 1995	
Penobscot R., ME, below head of tide	1.3 (F); 1 (N)	May 1992 -1993	(<i>n</i> = 68)	Blackwell et al. 1997	
Bays of Prince Edward Island	35.5 (B)	Breeding season, 1985	Stomach contents (<i>n</i> not reported)	Birt et al. (1987)	

Location	% in diet	Time	Assessment Method	Source	Variation
FLATFISHES (FLOUNDER SP.)					
Penobscot River, ME, below head of tide	6.9 (F)	Late Apr 1986-88	(n = 66)	Blackwell et al. 1997	Temporal
	0.3 (F)	May 1986-88	(n = 244)		
	10.5 (F)	Early June 1986	(n = 19)		
	0	Late Apr 1993	(n = 7)		
	1.7 (F)	May 1992-93	(n = 68)		
	0.85 (F)	Jun 1992-93	(n= 73)		
Eastern Shore rivers, Nova Scotia	4.8 (F); 2.1 (N)	Breeding season, 1979-80	Pellets, Boli (n not reported)	Milton et al. 1995	
Passamaquoddy Bay, New Brunswick	3.2 (R)	May 19-31 1986	Stomach contents (n =117)	Kehoe 1987 (in Cairns 1998)	
Estuary of St. Lawrence River, Québec	6.4 (F); 2.6 (N)	Jun 4-Jul 14, 1994	Boli (n = 125)	Rail and Chapeldaine 1998	Temporal; Spatial
	25.9 (F); 9.7 (N)	Jun 4-Jul 14, 1995	Boli (n = 301)		
Gulf of St. Lawrence River, St. Mary Is., Québec	10.1 (F); 2.2 (N)	Jun 28-Jul 20, 1995	Boli (n = 69)		
	9.3 (F); 2.0 (N)	Jun 28-Jul 20, 1996	Boli (n = 118)		
Bays of Prince Edward Island	18.1 (B)	Breeding season, 1985	Stomach contents (n not reported)	Birt et al. (1987)	

1 The study by Blackwell et al. (1997) reported frequency of occurrence and % volume of prey taxa for cormorants at and away from dams above the head of tide. It also reported the data in biweekly periods. To simplify the presentation of the data, we combined numbers reported at and away from dams above the head of tide, and we lumped numbers for biweekly periods (early May, late May, etc.). The categories we did not include (particularly “at” and “away from dams”) are important. For example, the ability of salmonids to detect and avoid predators has been shown to be negatively affected by dams (trying to pass them and associated migration delays) (Raymond 1979; Mesa et al. 1994), and cormorants were shown to select for use of areas adjacent to mainstem dams on the Penobscot River (Blackwell and Krohn 1997). The categories we didn’t include further indicate the influence of spatial and seasonal variation of prey populations on cormorant habitat use and diet composition.

Table 21. Atlantic Region: Occurrence of Estuarine / Diadromous fishes in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

B = biomass of taxon of interest as a percent of total sample biomass

F = percent of samples which contain at least one specimen of the taxon of interest

R = relative importance, sum of % of each prey species divided by number of stomachs analyzed

Location	% in diet	Time	Assessment Method	Source	Variation
ALEWIFE					
Penobscot River, ME, above head of tide	0 (F)	Late April 1986-88	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	2.4 (F)	May 1986-88	(n = 127)		
	2.6 (F)	May 1992-93	(n = 36)		
Penobscot River, ME, below head of tide	34.5 (F)	Late Apr 1986-88	(n = 66)		
	7.8 (F)	May 1986-88	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	14.3 (F); 1 (N)	Late Apr 1993	(n = 7)		
	9.4 (F); 11 (N)	May 1992-93	(n = 68)		
	10.35 (F); 7.5 (N)	Jun 1992-93	(n = 73)		
Passamaquody Bay, New Brunswick	12.70 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	
Eastern Shore rivers, Nova Scotia	3.4 (F); 1.6 (N)	Breeding season, 1979 – 1980	Pellets, Boli (n not reported)	Milton et al. 1995	
AMERICAN EEL					
Penobscot River, ME, above head of tide	0 (F)	Late April 1986 – 1988	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	13 (F)	May 1986-1988	(n = 127)		
	16.7 (F)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	3 (F)	Late Apr 1986 – 1988	(n = 66)		
	3.9 (F)	May 1986-1988	(n = 244)		
	10.5 (F)	Early June 1986	(n = 19)		
	0 (F)	Late Apr 1993	(n = 7)		
	4.3 (F)	May 1992-1993	(n = 68)		
	11.1 (F)	Jun 1992-1993	(n = 73)		
Eastern Shore rivers, Nova Scotia	3.7 (F); 2 (N)	Breeding season, 1979-80	Pellets, Boli (n not reported)	Milton et al. 1995	

Location	% in diet	Time	Assessment Method	Source	Variation
RAINBOW SMELT					
Penobscot River, ME, below head of tide	51.7 (F)	Late Apr 1986 – 1988	(n = 66)	Blackwell et al. 1997	Temporal
	7.3 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	0	Late Apr 1993	(n = 7)		
	6.5 (F); 2.5 (N)	May 1992-1993	(n = 68)		
	0	Jun 1992-1993	(n = 73)		
Lower Dunk River, Prince Edward Island	95.2 (N); 95.9 (B); 71.4 (F)	May 8-20 1993	Stomach contents (n = 14)	Hill et al. 1997	
	94.5 (B); 94 (F)	May 2-10 1995	Stomach contents (n = 151)		
Scales Pond ¹ , Prince Edward Island	89.2 (F)	May 8-10 1995	Stomach contents (n = 65)		
Estuary of St. Lawrence River, Québec	4.8 (F); 2 (N)	Jun 4-Jul 14, 1994	Boli (n = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	3 (F); 1.7 (N)	Jun 4-Jul 14, 1995	Boli (n = 301)		
Passamaquody Bay, New Brunswick	3.7 (R)	May 19-31, 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	
Eastern Shore rivers, Nova Scotia	0.8 (F); 0.2 (N)	Breeding season, 1979-1980	Pellets, Boli (n not reported)	Milton et al. 1995	
STICKLEBACK					
Scales Pond ¹ , PEI	75.4 (F)	May 8-10 1995	Stomach contents (n = 65)	Hill et al. 1997	Spatial
Passamaquody Bay, New Brunswick	2.6 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	
Eastern Shore rivers, Nova Scotia	0.4 (F); 0.2 (N)	Breeding season, 1979-1980	Pellets, Boli (n not reported)	Milton et al. 1995	
BLUEBACK HERRING					
Penobscot R., ME, below head of tide	0	Late Apr 1993	(n = 7)	Blackwell et al. 1997	Temporal
	6 (F)	May 1992-1993	(n = 68)		
	12 (F); 24 (N)	Jun 1992-1993	(n = 73)		

Location	% in diet	Time	Assessment Method	Source	Variation
HERRING					
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-1988	(n = 66)	Blackwell et al. 1997	Temporal
	1.4 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	14.3 (F); 1 (N)	Late Apr 1993	(n = 7)		
	11.95 (F); 10.25 (N)	May 1992-1993	(n = 68)		
	42.85 (F); 10 (N)	Jun 1992-1993	(n = 73)		
ATLANTIC TOMCOD					
Penobscot River, ME, below head of tide	1.5 (F)	Late Apr 1986-1988	(n = 66)	Blackwell et al. 1997	Temporal
	0 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	0	Late Apr 1993	(n = 7)		
	6 (F); 4 (N)	May 1992-1993	(n = 68)		
	12.6 (F); 3 (N)	Jun 1992-1993	(n = 73)		

1 Scales Pond is a freshwater impoundment on the Lower Dunk River.

Table 22. Atlantic Region: Occurrence of Freshwater fishes in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

F = percent of samples which contain at least one specimen of the taxon of interest

R = relative importance, sum of % of each prey species divided by number of stomachs analyzed

Location	% in diet	Time	Assessment Method	Source	Variation
SMALLMOUTH BASS					
Penobscot River, ME, above head of tide	36.1 (F)	Late April 1986-1988	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	8.1 (F)	May 1986-1988	(n = 127)		
	6.3 (F)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-1988	(n = 66)		
	0.3 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	42.9 (F)	Late Apr 1993	(n = 7)		
	0 (F); 0 (N)	May 1992-1993	(n = 68)		
	0 (F); 0 (N)	Jun 1992-1993	(n = 73)		
YELLOW PERCH					
Penobscot River, ME, above head of tide	0 (F)	Late April 1986-1988	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	9.53 (F)	May 1986-1988	(n = 127)		
	6.8 (F)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-1988	(n = 66)		
	0.3 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	28.6 (F)	Late Apr 1993	(n = 7)		
	0 (F, N)	May 1992-1993	(n = 68)		
	0 (F, N)	Jun 1992-1993	(n = 73)		
Passamaquody Bay, New Brunswick	0.10 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	
Eastern Shore rivers, Nova Scotia	0.8 (F); 1.1 (N)	Breeding season, 1979-1980	Pellets, Boli (n not reported)	Milton et al. 1995	

Location	% in diet	Time	Assessment Method	Source	Variation
PUMPKINSEED					
Penobscot River, ME, above head of tide	19.45 (F)	Late April 1986-1988	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	0.47 (F)	May 1986-1988	(n = 127)		
	0 (F)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-1988	(n = 66)		
	0.65 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	0 (F)	Late Apr 1993	(n = 7)		
	0 (F, N)	May 1992-1993	(n = 68)		
	2.6 (F)	Jun 1992-1993	(n = 73)		
CYPRINID SP.					
Penobscot River, ME, above head of tide	11.1 (F)	Late April 1986-1988	Stomach contents (n = 15)	Blackwell et al. 1997	Spatial
	9.8 (F)	May 1986-1988	(n = 127)		
	11.3 (F)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	1.5 (F)	Late Apr 1986-1988	(n = 66)		
	1.3 (F)	May 1986-1988	(n = 244)		
	0 (F)	Early June 1986	(n = 19)		
	0 (F)	Late Apr 1993	(n = 7)		
	0 (F, N)	May 1992-1993	(n = 68)		
	0.85 (F)	Jun 1992-1993	(n = 73)		

Table 23. Atlantic Region: Occurrence of Salmonids in the diet of Double-crested Cormorants.

N=number of specimens of the taxon of interest as a percent of all specimens in sample

B=biomass of taxon of interest as a percent of total sample biomass

F=percent of samples which contain at least one specimen of the taxon of interest

R=relative importance, sum of % of each prey species divided by number of stomachs analyzed

Location	% in diet	Time	Assessment Method	Source	Variation
ATLANTIC SALMON					
Penobscot River, ME, above head of tide	0 (F)	Late April 1986-1988	Stomach contents (n = 15)	Blackwell et al. 1997	Temporal; Spatial
	48.3 (F)	May 1986-1988	(n = 127)		
	70.1 (F); 15.5 (N)	May 1992-1993	(n = 36)		
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-1988	(n = 66)		
	48.26 (F)	May 1986-88	(n = 244)		
	5.3 (F)	Early June 1986	(n = 19)		
	0	Late Apr 1993	(n = 7)		
	22 (F); 27 (N)	May 1992-1993	(n = 68)		
0.85 (F); 0.5 (N)	Jun 1992-1993	(n = 73)			
Lower Dunk River, PEI	4.8 (N); 4.1 (B); 7.1 (F)	May 8-20 1993	Stomach contents (n = 14)	Hill et al. 1997	
	5.3 (F); 4.2 (B);	May 2-10 1995	Stomach contents (n = 151)		
Passamaquody Bay, New Brunswick	0.10 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	
BROOK TROUT					
Lower Dunk River, PEI	1.3 (F); 1.2 (B)	May 2-10 1995	Stomach contents (n = 151)	Hill et al. 1997	
SALMONIDS (mostly ATLANTIC SALMON)					
Eastern Shore rivers, Nova Scotia	13.8 (F) ¹	Apr-Jun 1980-1981	Stomach contents (n = 65)	Milton et al. 1988	
	39.8 (N)	Apr-Jun 1986-1987	Stomach contents (n = 83)		

1 Only wild salmon stocks were available in 1980-1981 because hatchery stocks were not released in sampled rivers (Milton et al. 1995).

Table 24. Atlantic Region: Occurrence of Pelagic fishes in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

F = percent of samples which contain at least one specimen of the taxon of interest

Location	% in diet	Time	Assessment Method	Source	Variation
AMERICAN SAND LANCE					
Estuary of St. Lawrence R., Québec	0 (F); 0 (N)	Jun 4-Jul 14, 1994	Boli (<i>n</i> = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	6.3 (F); 6.5 (N)	Jun 4-Jul 14, 1995	Boli (<i>n</i> = 301)		
Gulf of St. Lawrence R., St. Mary Is., Québec	68.1 (F); 85.3 (N)	Jun 28-Jul 20, 1995	Boli (<i>n</i> = 69)		
	52.5 (F); 59.3 (N)	Jun 28-Jul 20, 1996	Boli (<i>n</i> = 118)		
E. Shore rivers, NS	5.2 (F); 6.2 (N)	Breeding season, 1979-1980	Pellets, Boli (<i>n</i> not reported)	Milton et al. 1995	
ATLANTIC HERRING					
E. Shore rivers, NS	7.8 (F); 4.6 (N)	Breeding season, 1979-1980	Pellets, Boli (<i>n</i> not reported)	Milton et al. 1995	
Estuary of St. Lawrence R., Québec	1.6 (F); 0.4 (N)	Jun 4-Jul 14, 1994	Boli (<i>n</i> = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	14.3 (F); 4.8 (N)	Jun 4-Jul 14, 1995	Boli (<i>n</i> = 301)		
Gulf of St. Lawrence R., St. Mary Is., Québec	1.4 (F); 0.2 (N)	Jun 28-Jul 20, 1995	Boli (<i>n</i> = 69)		
	7.6 (F); 1.4 (N)	Jun 28-Jul 20, 1996	Boli (<i>n</i> = 118)		
CAPELIN					
Estuary of St. Lawrence R., Québec	48.8 (F); 41.9 (N)	Jun 4-Jul 14, 1994	Boli (<i>n</i> = 125)	Rail and Chapdelaine 1998	Temporal; Spatial
	47.5 (F); 53.8 (N)	Jun 4-Jul 14, 1995	Boli (<i>n</i> = 301)		
Gulf of St. Lawrence R., St. Mary Is., Québec	7.2 (F); 1.5 (N)	Jun 28-Jul 20, 1995	Boli (<i>n</i> = 69)		
	55.9 (F); 30.9 (N)	Jun 28-Jul 20, 1996	Boli (<i>n</i> = 118)		
ATLANTIC MENHADEN					
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1986-88	(<i>n</i> = 66)	Blackwell et al. 1997	Temporal
	0 (F)	May 1986-88	(<i>n</i> = 244)		
	0 (F)	Early June 1986	(<i>n</i> = 19)		
	0 (F)	Late Apr 1993	(<i>n</i> = 7)		
	5.6 (F)	May 1992-93	(<i>n</i> = 68)		
	15.3 (F)	Jun 1992-93	(<i>n</i> = 73)		

Table 25. Atlantic Region: Occurrence of Crustaceans in the diet of Double-crested Cormorants.

N = number of specimens of the taxon of interest as a percent of all specimens in sample

F = percent of samples which contain at least one specimen of the taxon of interest

R = relative importance, sum of % of each prey species divided by number of stomachs analyzed

Location	% in diet	Time	Assessment Method	Source	Variation
SAND SHRIMP					
Penobscot River, ME, below head of tide	0 (F)	Late Apr 1993	(n = 7)	Blackwell et al. 1997	Temporal
	43.3 (F)	May 1992-93	(n = 68)		
	56.7 (F)	Jun 1992-93	(n = 73)		
SHRIMP SP.					
Penobscot River, ME, below head of tide	17.2 (F)	Late Apr 1986-1988	(n = 66)	Blackwell et al. 1997	Temporal
	7.8 (F)	May 1986-88	(n = 244)		
	63.2 (F)	Early June 1986	(n = 19)		
	0(F)	Late Apr 1993	(n = 7)		
	5.6 (F)	May 1992-93	(n = 68)		
	0 (F)	Jun 1992-93	(n = 73)		
Estuary of St. Lawrence R., Québec	5.6 (F); 1.3 (N)	Jun 4-Jul 14, 1994	Boli (n = 125)	Rail and Chapdelaine 1998	
	5 (F); 2.3 (N)	Jun 4-Jul 14, 1995	Boli (n = 301)		
Gulf of St. Lawrence R., St. Mary Is., Québec	4.3 (F); 1.2 (N)	Jun 28-Jul 20, 1995	Boli (n = 69)		
	3.4 (F); 0.6 (N)	Jun 28-Jul 20, 1996	Boli (n = 118)		
OTHER CRUSTACEANS					
Estuary of St. Lawrence River, Québec	9.6 (F); 3.8 (N)	Jun 4-Jul 14, 1994	Boli (n = 125)	Rail and Chapdelaine 1998	Temporal
	0.3 (F); 0.2 (N)	Jun 4-Jul 14, 1995	Boli (n = 301)		
Passamaquody Bay, New Brunswick	6.3 (R)	May 19-31 1986	Stomach contents (n = 117)	Kehoe 1987 (in Cairns 1998)	

PART IV. IMPACTS TO VEGETATION AND COLONIAL WATERBIRDS

Through their normal nesting activities, DCCOs have been shown to impact vegetation on a local level. These impacts affect not only plants, but can affect the animals that depend on them, particularly other colonial waterbirds. Because these impacts have undoubtedly occurred throughout the evolutionary histories of co-occurring animals and plants, such interactions should be viewed as a natural phenomenon within this particular system. However, in human-altered ecosystems where habitat is limited, these interspecific dynamics may prove deleterious to some species. Additionally, humans may be impacted when DCCOs kill trees valued for aesthetics or timber.

Generally, impacts are most concentrated at breeding colonies, but can also occur at roosting sites. Concern over impacts to vegetation and other colonial waterbirds has been great enough to initiate control programs on islands in the eastern basin of Lake Ontario, Lake Champlain, VT (USFWS 1999a,b), the St. Lawrence River Estuary (Bédard et al. 1995), and Oneida Lake (R. Miller, pers. comm.). Additionally, there has been much speculation about the impacts of DCCOs on other avian species (see also *Managing Impacts to Vegetation and Habitat* in PART V. MANAGEMENT OPTIONS and Appendix I. Perceived Impacts Associated With Double-crested Cormorants). However, only a few studies have been conducted to assess the actual affects DCCO breeding ecology and behavior have on vegetation and other colonial waterbirds.

IMPACTS TO VEGETATION

The most important impact to vegetation ascribed to cormorants is the lethal effect their acidic guano has on ground vegetation and eventually nest trees. This effect has been well documented at Lake Ontario breeding colonies, including Little Galloo Island, New York, Hamilton Harbour, Ontario (Weseloh and Ewins 1994; Moore et al. 1995; Weseloh and Collier 1995), and on islands in the St. Lawrence River Estuary (Bédard et al. 1995). J. Bédard (pers. obs. in Bédard et al. 1995) reported that DCCO guano deposited on forest soil causes irreversible damage to trees in < 3 years. Damage to island property was reported as “ecologically measurable” (however, measurements were not provided by Bédard et al. 1995). Moore et al. (1995) reported that DCCOs began nesting at Hamilton Harbour in 1984, and beginning in 1986, they nested mostly in top branches of cottonwood trees at Pier 27. Since that time, trees have gradually died; they were slowly defoliated and by 1993, only 24 % remained alive. Deposition of DCCO guano is presumed to have killed them, and the loss of all cottonwood trees at this site is inevitable.

Cormorants also directly destroy vegetation by stripping leaves from trees, and the combined weight of the birds and their nests can break branches (Weseloh and Ewins 1994; Weseloh and Collier 1995). Additionally, in some areas, such as the islands of western Lake Erie, the vegetation is of unusual significance or rare, and large DCCO colonies in such areas may significantly impact or destroy plants and or plant communities of special interest (Weseloh and Collier 1995). The loss of vegetation can also lead to a reduction in general species richness at colony sites (Chapdelaine and Bédard 1995).

Through their nesting and roosting habits DCCOs have undoubtedly always had some impact on vegetation. However, human induced environmental changes, such as interruptions in

the natural flow of river systems, dredging operations, erosion, deforestation, etc., may magnify these impacts as natural areas and breeding bird habitat become more limited (e.g., Kirsch 1995).

IMPACTS TO AVIAN SPECIES

Two potential effects of DCCOs on other colonial waterbirds is site competition and habitat degradation (Shieldcastle et al. in press; Weseloh and Ewins 1994; Moore et al. 1995). Birds such as Black-crowned Night-Herons (BCNH) may be affected directly through competition for nests and nest-sites (Moore et al. 1995; Jarvie et al. 1999), but thus far this has not been documented. In some areas, DCCOs may arrive at islands earlier than other species and usurp quality nest sites (USFWS 1999b; Jarvie et al. 1999). However, most impacts to other birds appear to occur indirectly through habitat degradation. Because DCCOs defoliate trees, the trees become less attractive as nest sites to certain species, e.g. BCNH, which prefer concealed areas for nesting (Moore et al. 1995).

In the mid-to-late 1980s at Hamilton Harbour, Ontario, DCCOs and BCNHs were observed to nest in the same trees until the trees began losing their leaves and dying. The BCNHs eventually moved to sandbar willow habitat, while the DCCOs remained in the trees. During the time when DCCOs and BCNHs nested in the same trees, they were distinctly separated spatially; of the DCCO nests in trees in 1993, 98 % were built > 2 m above the ground, while 94 % of BCNH nests occurred in elevations < 2 m above the ground. Guano and debris falling on BCNHs and their nests from DCCO nests positioned higher in the same tree may have also precipitated the observed habitat shift (Moore et al. 1995). Habitat surveys in 1993 also indicate that DCCOs and BCNHs were coexisting in the colony by utilizing distinctly different vegetation types. However, as sandbar willows grow larger they may become suitable for DCCOs, which may lead to the eventual displacement of BCNHs at this site; DCCOs have displaced BCNHs from nesting sites in shrubs at other Lake Ontario colonies (Moore et al. 1995).

On West Sister Island, Lake Erie, DCCOs were first documented nesting in 1992, and the first habitat damage to the island was noted in 1995; major leaf loss occurred on trees with cormorant nests. Entire trees were nearly defoliated along the island edge where heavy roosting by DCCOs occurs. Increases in DCCO numbers on the island have corresponded with declines in BCNHs and Great Blue Herons (GRBH). Shieldcastle et al. (in press) predict that once trees die and fall, DCCOs will likely become ground nesters, and GRBHs will be displaced from the island. Moore et al. (1995) suggest that a switch to ground nesting by DCCOs at Hamilton Harbour may also detrimentally impact Herring Gulls already nesting there, but do not explain how.

Tree loss by itself appears to have the greatest effect on subcanopy nesters, such as BCNHs. On West Sister Island, however, a factor that complicates assessment of DCCO impact on BCNHs is that habitat loss for BCNHs also appears to be occurring through successional change. As canopy height increases there appears to be a negative effect on nesting BCNHs; declines in this species actually began in 1991, the year before DCCOs were first observed nesting on the island (Shieldcastle et al. 1999). At Tommy Thompson Park, Lake Ontario, increasing numbers of DCCOs correspond with decreasing numbers of BCNHs on peninsulas A and B (Jarvie et al. 1999). However, declines in some individual BCNH subcolonies, especially on peninsula B, appear to have occurred in areas distinct from those where DCCOs were increasing.

While the ability of some species to switch nesting habitat may facilitate co-existence in a colony, switching nesting habitat may entail costs. At Hamilton Harbour, several factors may

affect BCNH reproductive success; all are related to the vulnerability of present nesting sites in sandbar willows and on the ground (Moore et al. 1995).

Long-term observations of colonial waterbirds suggest that colonial waterbird colonies are dynamic, though some species shift colony sites less frequently than others (Ferren and Myers 1998b; F. Cuthbert pers. obs.). Colonies of most species experience habitat degradation due to deposition and accumulation of guano, causing cyclical patterns of colonization and abandonment, followed by a period during which a colony site will lie “fallow”, and then later be re-colonized (Ferren and Myers 1998b). A pattern of use, tree die off due to guano deposition, abandonment and eventual re-colonization was documented for DCCOs at a colony at Great Lake, North Carolina (Pearson et al. 1959; see PART II. POPULATION DATA AND TRENDS, North Carolina summary). As this cyclical pattern of use and abandonment occurs, impacts to other colonial waterbirds from DCCOs should be insignificant, especially when viewed over longer time periods and wider geographical scales. However, if habitat and alternative quality sites are limited, impacts may be significant.

In conclusion, observations suggest DCCOs may displace other waterbirds from individual colonies; however, in some situations where this appeared to be the case, further investigation identified other factors (e.g. vegetational succession) that may be related to shifts in nesting patterns or local declines, especially when these declines begin in areas prior to or disjunct from cormorant nesting. While displacement may occur locally, no studies have demonstrated regional or population level impacts on other colonial waterbirds. Even at the colony level changes in parameters such as reproductive success of birds displaced by DCCOs have not been measured. More detailed studies are needed to determine the exact nature of competitive interactions between DCCOs and other birds. Additionally, the effects of interactions on nesting habitat selection and breeding success of other species needs to be examined (Moore et al. 1995).

NEWCASTLE DISEASE

(summarized from Kuiken 1999 unless otherwise noted)

Newcastle disease virus is the causative agent of Newcastle disease, an important disease for the poultry industry because it causes devastating epidemics and has far-reaching effects on trade in poultry products. The disease can affect virtually all avian species, and is commonly passed through bird fecal material, though it can also be passed through other sources, such as humans who have been handling infected birds. In Double-crested Cormorants, the disease was recognized for the first time in 1975 (Québec, St. Lawrence River Estuary), and may cause high mortality of juvenile DCCOs. Typically, birds infected with the disease will have wing or leg paralysis, loss of balance, and inflammation of the brain and spinal cord.

Between 1990 – 1997, the disease occurred in DCCOs across the full east-west expanse of their range in North America, occurring at least four times. In 1990, the disease was associated with high mortality of juveniles in several breeding colonies across western Canada. In 1992, a more widespread epidemic occurred, and DCCOs from the interior population were broadly affected. Weseloh and Collier (1995) estimated that during this outbreak up to 30 % of young DCCOs were killed at several colonies, while Kuiken (1999) cites rates ranging from 10 – 90 % mortality of juveniles per breeding colony. Outbreaks in DCCOs occurred again in 1995 and possibly 1996 (Wildlife Health Centre Newsletter 1997).

Prior to 1997, all recognized occurrences had been in populations east of the Rocky Mountains, although antibodies to Newcastle Disease were found in eggs from nests in British Columbia in 1993 (Wildlife Health Centre Newsletter 1997). In May 1997, personnel at the Salton Sea NWR, east of San Diego, reported approximately 1,600 dead nestlings and fledglings in a mixed colony of DCCOs and Caspian Terns. Most of the dead were DCCOs. A team from the National Wildlife Health Center secured samples for diagnostic examination, and Newcastle Disease virus was isolated from the affected tissues (Wildlife Health Centre Newsletter 1997). In the same year, the disease was also diagnosed in juvenile DCCOs from colonies at the mouth of the Columbia River (Oregon) and from Great Salt Lake (Utah) (L.C. Glaser, pers. comm. in Kuiken 1999). Mortality of juveniles in these 3 areas ranged from “not normal” to > 90 %.

There have been no reports of extensive mortality to other wild birds cohabiting with DCCOs due to Newcastle disease. However, small numbers of American White Pelicans and Ring-billed Gulls sharing colony sites with infected DCCOs have been documented as infected with the disease, and in 1992, the widespread epidemic was associated with high mortality in American White Pelicans and gulls (*Larus spp.*). In 1995, the virus was also found in a Caspian Tern in Ontario in the same area where infected DCCOs occurred. In 1992, Newcastle disease may also have been passed from infected DCCOs to commercial range turkeys in North Dakota; these turkeys were located < 7 km from a DCCO colony known to be affected with the disease.

In conclusion, infection in other birds from DCCOs with Newcastle disease has not been clearly documented to be a significant problem; however, the potential for DCCOs to contract and spread the disease to both wild and domestic birds exists.

PART V. MANAGEMENT OPTIONS

The J. Paul Getty Museum, Los Angeles



Figure 88
Detail of *Hunting on the Lagoon* by Vittore Carpaccio
ca. 1490-1495, oil on panel, 75.4 x 63.8 cm.

INTRODUCTION

Cormorant management to reduce conflicts with human interests has taken place in the eastern hemisphere for centuries. Possibly one of the earliest records of cormorant control may be the late fifteenth century painting by Vittore Carpaccio, “Hunting on the Lagoon,” (Figure 88). According to Conniff (1991), the scene depicts Venetian archers in lagoon fish enclosures shooting at cormorants with pellets. It is not certain whether these archers are trying to prevent depredations to fish stocks or are merely hunting cormorants for food or pleasure, and some question whether the archers may actually be shooting at fish rather than cormorants (J. Hatch, pers comm.). However, the pellet shot by the foremost archer and about to hit a cormorant in the

head is suggestive of lethal intent towards the cormorants. More solid evidence of cormorant control appears a century later (1599) when Queen Elisabeth I brandished cormorants as “pests of the crown”; in 1603, the last year of her life, she placed a bounty on them as well (Ludwig et al. 1995). Here in North America cormorants have long been accused of negatively impacting fish populations (e.g. Wood 1634) and have been persecuted because of their fish-eating habits (Lewis 1929; Mendall 1936). However, few details about cormorant management to reduce impacts to fisheries prior to 1900 exist. Although cormorants and their eggs were eaten by Native Americans and early settlers, and nestlings were used as food for dogs and farm-raised foxes (Lewis 1929), we found no evidence of active management to increase or maintain cormorant numbers prior to 1900.

As DCCOs began to recover from 19th Century declines during the first half of the 20th Century (see PART II. POPULATION DATA AND TRENDS), cormorant management focused on reducing impacts to fisheries, although impacts to timber resources may also have been of concern (e.g. McLeod and Bondar 1953). Documented management efforts from this period into the early 1950s included official government programs and private citizen actions. The scale of these efforts varied considerably, from small-scale, uncoordinated activities of individual fishermen to Maine’s multi-year program along its entire coast.

In response to declines in DCCO populations from the 1950s through the early 1970s management began to include efforts to increase cormorant numbers. The species was listed as endangered in Wisconsin (1972) and Illinois (1960) and as “in need of management” in Tennessee (1976). Artificial nesting structures were erected in Wisconsin and Illinois. As a result of many factors (see PART II. POPULATION DATA AND TRENDS) DCCO populations soon began to recover, and efforts to control the species resumed. Conditions had changed, however, resulting in new restrictions to management and new impacts of concern. DCCOs received federal protection under the Migratory Bird Treaty Act in 1972, and lethal control activities by individual citizens thereafter required special permits from USFWS. Aquaculture in the southern U.S. grew considerably in the late 1960s and early 1970s while DCCO populations were very low, and as DCCO populations recovered, predation at aquaculture facilities challenged the economic assumptions under which the industry had initially evolved (Hodges 1989; Jackson and Jackson 1995). Throughout the species’ breeding range, alteration of vegetation from DCCO nesting activities led to concerns of impacts to species already threatened by human activities as well as impacts to private property.

Currently in the U.S. all lethal take requires permits from the USFWS, except in states under the Depredation Order. Depredation permits can be obtained to prevent economic impacts or impacts to endangered, threatened or species of conservation concern. Non-lethal harassment of birds depredating or about to depredate does not require permits. In some states additional permits are required for lethal and / or non-lethal control, including harassment.

In this section we used published and unpublished literature to review and describe the techniques available for managing DCCO impacts, we assess numerical and economic effectiveness (when data are available), and we provide case studies where appropriate. We separated management options into three main categories based on DCCO impacts: 1) aquaculture depredations, 2) alteration of vegetation and habitat, and 3) fisheries depredations. We also asked survey recipients (see SURVEY RESULTS, below) to provide information on control efforts undertaken in their state or province. This information is summarized in Table 28 and discussed at the end of this section. Throughout this section we use the term *control* to refer to both lethal and non-lethal management efforts to reduce cormorant impacts.

Table 26. Sources of information on DCCO control at aquaculture facilities.

Citation	Dates of study	Geographical range	Aquaculture facilities	Depredating Species	Type of study/paper
Lagler 1939	1937-8	United States	Fish hatcheries	All piscivores	Survey
Brémond 1980	NA	Global	Gen. agriculture	Avian	Theory & lit. rev.
Inglis 1980	NA	Global	Gen. agriculture	Avian	Theory & lit. rev.
Krebs 1980	NA	Global	Gen. agriculture	Avian	Literature review
Slater 1980	NA	Global	Gen. agriculture	Avian	Theory & lit. rev.
Barlow & Bock 1984	1979-80	Australia	Farm dams	Cormorants	Field study
Moerbeek et al. 1987	1981-3	Netherlands	Fish farm (carp)	GRCO	Field study
Parkhurst et al. 1987	1984	U.S. (eastern)	Hatch & nurseries	All piscivores	Survey
Schramm et al. 1987	NA	Florida	Gen. aquaculture	Avian piscivores	Literature review
Hanebrink & Byrd 1989	NA	Arkansas	Gen. Aquaculture	Avian piscivores	Survey
Hodges 1989	1987-8	Mississippi	Catfish farms	Avian piscivores	Field study
Stickley & Andrews 1989	1988	Mississippi	Catfish farms	Avian piscivores	Survey
Littauer 1990a,b	NA	U.S. (southern)	Gen. aquaculture	Avian piscivores	Info brochure
Conniff 1991	NA	U.S. (southern)	Gen. aquaculture	DCCOs	Magazine article
Brugger 1995	NA	Florida	Gen. aquaculture	DCCOs	Survey
Duffy 1995	NA	North America	Gen. aquaculture	DCCOs	Symposium rev.
Erwin 1995	NA	North America	Gen. aquaculture	DCCOs	Literature review
Glahn & Stickley 1995	NA	Mississippi	Catfish farms	DCCOs	Literature review
Keith 1995	NA	Canada	Gen. aquaculture	DCCOs	Lit. & policy rev.
Mott & Boyd 1995	NA	Southeast U.S.	Gen. aquaculture	DCCOs	Literature review
Mott et al. 1995	1992	Mississippi	Catfish farm	DCCOs	Field study
Nisbet 1995	NA	North America	Gen. Aquaculture	DCCOs	Symposium rev.
Price & Nickum 1995	NA	North America	Gen. aquaculture	Avian piscivores	Literature review
Stickley & King 1995	1992	Mississippi	Catfish farm	DCCOs	Field study
Stickley et al. 1995	1991	Mississippi	Catfish farm	DCCOs	Field study
Thompson et al. 1995	NA	Texas	Gen. aquaculture	DCCOs	Lit. rev. & field s.
Pitt & Conover 1996	1993-4	U.S. (western)	Fish hatcheries	All piscivores	Observ. & survey
Spencer 1996	1992-4	Georgia	Fish hatcheries	DCCO, other avian	Survey
Bregnballe et al. 1997	NA	Europe	Gen. aqua & fish.	GRCO	Population model
Littauer et al. 1997	NA	U.S. (southeast)	Gen. aquaculture	Avian piscivores	Info brochure
Mott & Brunson 1997	NA	Mississippi	Catfish farms	Avian piscivores	Literature review
Simmonds et al. 1997	1992-4	Oklahoma	Catfish farms	DCCOs	Field study/survey
Depr. Order; Trapp 1998	NA	U.S. (southeast)	Gen. aquaculture	DCCOs	Literature review
Gottfried 1998	1996-7	Louisiana	Catfish farms	DCCOs	Field study
Keller et al. 1998	NA	Bavaria, Germany	Gen. aqua & fish.	GRCOs	Field study
Wywalowski 1998, 1999	1997	U.S., primarily S	Catfish farms	All piscivores	Survey
Frederiksen & Lebreton; Frederiksen et al. 1999	NA	France	General aqua. & fisheries	GRCOs	Population model
Glahn et al. 1999	NA	U.S. (southern)	Gen. aquaculture	DCCOs	Literature review
Hatch & Weseloh 1999	NA	North America	Gen. aquaculture	DCCOs	Literature review
Keller 1999a	1992-6	Bavaria, Germany	Gen. aquaculture	GRCOs	Field study
Keller 1999b	NA	Bavaria, Germany	Gen. aqua. & fish	GRCOs	Field study
McKay et al. 1999	NA	England & Wales	Gen. aqua. & fish.	Avian piscivores	Field s. & lit. rev.
Reinhold & Sloan 1999	NA	Mississippi	Gen. aqua./catfish	DCCOs	Literature review

MANAGING IMPACTS TO AQUACULTURE

Aquaculture fisheries impacted by DCCO depredations include food fishes, game fishes and bait fishes; exotic fish aquaculture does not appear to be negatively impacted by DCCOs (Brugger 1995; Price and Nickum 1995). Of the three impacted categories, catfish farming, especially in the Mississippi Delta region, has received the most discussion in the scientific literature. Numerous techniques have been devised and tested for reducing DCCO depredations at catfish farms and are reviewed below.

Estimates of the economic effectiveness of some of these techniques have been made, and in some cases these estimates may be helpful for comparing effectiveness between techniques. However, because net impacts of DCCOs to aquaculture remain unknown (see PART III: DIET) it is not possible to determine which techniques are economically effective at recovering losses due to cormorants.

In a broad review of scientific literature such as this, difficulties arise in attempting to summarize papers that not only differ temporally and geographically (important with changing DCCO populations), but also vary in application of techniques and even definitions of “success.” For example, some investigators have promoted rational economic assessments of impacts and control techniques to determine whether and to what extent control is necessary (Parkhurst et al. 1992; Thompson et al. 1995; Littauer et al. 1997; Reinhold and Sloan 1999). In contrast, others may not consider control successful unless all cormorants are removed from the ponds, regardless of economics. Variability also arises in surveys in which aquaculturists are asked to qualitatively rate the effectiveness of control techniques based on their own definitions of success (e.g., Lagler 1939; Parkhurst et al. 1987; Hanebrink and Byrd 1989; Sticklely and Andrews 1989; Brugger 1995; Spencer 1996; Wywalowski 1998, 1999). To facilitate comparisons among studies cited in this section, general information about each has been summarized in Table 26.

1) Exclusion Devices: Physical and Functional Barriers

The two most commonly discussed devices to exclude cormorants are nets and wires. Nets act as physical barriers to cormorants; overhead wires act as functional barriers (i.e. do not exclude cormorants, but make it difficult for them to land on and take off from ponds). Floating ropes have been tested as cheaper alternatives to overhead wires. Additional exclusion devices / techniques include underwater structures that hinder cormorant access to fish and tall trees around ponds that may limit cormorant ability to take flight. Although exclusion devices are currently in use at fish hatcheries, Wywalowski (1998, 1999) does not report any catfish producers using exclusion/barrier systems in 1996.

Netting

Numerically, nets are the most effective technique for preventing cormorant depredations: cormorants that are physically prevented from accessing ponds cannot prey on fish within (Littauer et al. 1997; Mott and Brunson 1997). Netting and similar physical exclusion devices have been used at fish hatcheries since at least the 1930s (Lagler 1939), and continue to be used successfully at aquaculture facilities with small ponds (Parkhurst 1987; Brugger 1995; Pitt and Conover 1996). To be completely effective, however, the edges of the nets must extend to the ground surrounding ponds. Cormorants have learned to land on pond levies and walk into

the water to bypass netting in Georgia (Spencer 1996) and wire systems in Australia (see Wires, below; Barlow and Bock 1984).

Despite their great potential for reducing predation, nets have rarely been used by catfish farmers (Stickley and Andrews 1989; Reinhold and Sloan 1999; Wywialowski 1999). The reasons given are that 1) large size of ponds typically constructed for raising catfish makes netting systems prohibitively expensive, and 2) levies between ponds are typically too narrow to hold both net support structures and farm machinery (Lagler 1939; Schramm et al. 1987; Hanebrink and Byrd 1989; Brugger 1995; Mott and Boyd 1995; Mott et al. 1995; Price and Nickum 1995; Littauer et al. 1997; Depredation Order, Trapp 1998). Price and Nickum (1995) state that use of multi age-class stock and other economic arguments precludes reconfiguring of ponds to allow netting and wire systems.

Sizes of catfish ponds (in Mississippi) have been described by Price and Nickum (1995) as typically ranging from 4 – 14 ha (average = 5 ha), by Glahn and Stickley (1995) as averaging 8 ha and ranging in depth from 1 to 2 m (with the typical farm having approximately 40.5 ha of ponds total), and by Mott and Brunson (1997) as averaging 5 – 6 ha (with the average Delta farm totaling 175 ha of water). Littauer et al. (1997) state that typical catfish ponds are 4.9 to 6.1 ha, and at an estimated \$0.22 / ft² it would cost approximately \$1 million to enclose a 40.5 ha farm. Conniff (1991) estimates the cost of covering an 8 ha pond with nets at \$22,000. Brugger (1995) cites an estimate of \$12,141 / ha (\$30,000 / acre) for netting in Florida, and concludes that hazing and dispersal are more economically effective.

The inconvenience of working around netting systems during maintenance and harvest activities is another concern (Lagler 1939; Brugger 1995; Mott and Boyd 1995; Littauer et al. 1997); Lagler (1939) discussed the possibility that increased labor demands incurred with netting systems might be offset by reduced harassment and other control activities.

No economic analyses appear to have explicitly compared costs of erecting netting over catfish ponds to potential losses to cormorants. Additionally, the data and calculations used in the cost estimates above were not provided. Therefore, it is not possible to determine whether this control technique has been sufficiently investigated to make conclusions about its lack of economic effectiveness. An analysis that more thoroughly compares costs of various materials and configurations, economies of scale, adaptation to (and of) maintenance and harvest activities, net losses at harvest due to predation, and various levels of government subsidies is needed to determine if this technique deserves more consideration.

Wires

Because cormorants require relatively long distances of open water to take flight (Duffy 1995) wires (or other filaments) over aquaculture ponds may make them less attractive feeding locations, thereby reducing predation. As with netting, wire systems have been used at fish hatcheries since the 1930s (Lagler 1939). Surveys of aquaculture facilities report moderate effectiveness of wire systems (Pitt and Conover 1996; Spencer 1996). Several studies tested use of wire systems to protect commercial aquaculture facilities from cormorant predation. Moerbeek et al. (1987) studied several wire configurations in the Netherlands to protect fish from GRCOs, and found that in general wire systems were successful at deterring large flocks from landing, but that individual birds learned to fly between the lines. McKay et al. (1999) reported wiring systems may be more effective against GRCOs in England and Wales when alternative foraging sites are available. Davis (1990, as cited by Mott and Boyd 1995) also reported that various wiring systems used in Texas deterred large flocks of DCCOs and

Neotropic Cormorants, but individual birds were less likely to be deterred. Simmonds et al. (1997) found twine stretched over narrow ponds to be effective against most cormorants in Oklahoma, but some birds learned to fly under the ends of the twine that were raised to counter sagging in the middle. Brugger (1995), on the other hand, reported that wire grids were effective in Florida only if *few* birds were present.

Problems with overhead wire systems include technical and economic problems of construction over large catfish ponds (see Netting, above, for sizes), access limitations to farmers, and hazards to non-target avian species such as Osprey and swallows (Mott and Boyd 1995). Birds may also learn to bypass wire barriers by landing on levies and walking into ponds (Barlow and Bock 1994) or by learning to fly between the wires (Moerbeek et al. 1987).

Spencer (1996) and Mott and Boyd (1995) both cite Littauer (1990b) to estimate cost of covering a 40.5 ha farm with wires at more than \$100,000 (at \$0.025 / ft²), or \$2500 / ha. Smaller ponds may be protected more economically: according to Littauer et al. (1997), stringing wire over one 3.7 ha pond cost \$66 / ha and required 15 working days to install, while smaller ponds (up to 0.9 ha) could be protected by wires for as little as \$6 / ha and required only 3 working hours to install. As was the case for netting, systematic analyses of costs and benefits of wire systems to reduce DCCO impacts have not been performed. In Bavaria, Germany, carp ponds of up to 4 ha were successfully protected from GRCO predation using various wire systems; benefit:cost analyses found some of these systems to be cost effective, and others became cost effective when subsidies by the Bavarian government paid for up to 40 % of costs (Keller 1999a).

The economic effectiveness of this technique will depend on two factors: 1) proportion of individual birds that adapts to bypassing the wires and 2) availability of alternative foraging sites (see: 5) Provision of Alternative Prey and Foraging Sites, below). The former has received little discussion in the literature. The latter will depend on how widely this technique is utilized and the proximity of individual farms to either natural or artificial alternative foraging areas.

Floating Ropes

Rather than suspending wires over ponds, ropes with attached floats may be strung across the surface of the water to produce a similar impediment to landing and takeoff. This technique is less expensive and easier to install than wire systems. Mott et al. (1995) found that floating ropes strung parallel to each other 15 – 17 m apart dramatically reduced the number of cormorants entering ponds (96 % reductions for each of two test ponds). A few birds appeared to habituate to the ropes; however, their numbers remained considerably lower than pre-treatment levels, and the addition of Mylar balloons appeared to reduce the number of habituated birds. At approximately \$100 / ha, the rope system was considerably cheaper than suspended wire systems of similar size (see above), and was expected to quickly pay for itself. These results should be interpreted with caution, however, as a maximum of 2 ponds was treated within a facility of 21 adjacent ponds; it is unknown whether the decrease in DCCO activity would be comparable if all ponds in a facility or region were similarly treated. Still, it should be noted that the two ponds selected for treatment were selected because they had the highest cormorant foraging rates prior to treatment, and application of the floating ropes was successful at shifting cormorant foraging to apparently less preferable sites. As with wire systems and harassment, proximity of alternative foraging sites will likely improve the effectiveness of this technique.

Although floating ropes have not been discussed in recent reviews of cormorant management techniques (Mott and Boyd 1995; Reinhold and Sloan 1999), this technique shows promise and should receive further investigation at larger scales than previously tested.

Other Flight Inhibitors

Duffy (1995) suggested constructing a “small-scale equivalent of the barrage balloons that the British used in World War II to discourage low-level bombers,” as another technique for making take-off from ponds difficult. This concept has not been tested by itself, but only as part of the floating rope system (Mott et al. 1995). Pond size and surrounding tree height may also impact cormorant ability to take off from the water, making certain ponds unattractive to DCCOs. Some investigators have suggested that such factors should be included in the construction of new facilities (Simmonds et al. 1997).

Underwater Exclusion Devices

Barlow and Bock (1984) suggest that cormorant depredations in remote fish dams (stock watering ponds) in Australia may be reduced by placing fish refugia in the dams, either to hinder cormorant predation or to give fish a place to hide during the day when most cormorant predation occurs; however, they did not test this technique. In general, underwater habitat modification has received little attention (Hatch and Weseloh 1999). In the Netherlands a small exclusion device was placed in the corner of a 10 ha pond in which young carp were reared until they were large enough to escape cormorant (*P. carbo*) predation and were released; the technique was “promising” (Moerbeek et al. 1987).

Gottfried (1998) tested the effectiveness of submerged nets as escape cover for catfish in aquaculture ponds in Louisiana. Experimental design included both switching treatment and control ponds to eliminate pond effects and quantification of DCCO predation success rates using behavioral observations. Results showed that DCCOs were equally successful at catching catfish in control and treatment ponds; further research on this technique has not occurred. No data are available to explain why these nets did not decrease DCCO predation rates, but V. Wright (pers. comm.) speculated that hatchery-raised catfish may lack behavior patterns needed to utilize escape cover.

Experimental testing of fish refuges in England and Wales demonstrated that these devices may reduce fish predation and injury rates by GRCOs (McKay et al. 1999).

2) Non-lethal Harassment at Aquaculture Facilities

There are numerous techniques and devices that have been used to frighten cormorants from aquaculture facilities to reduce predation. However, systematic analyses of these techniques are limited; most available data are anecdotal or from aquaculturist surveys. These techniques are reviewed within three categories: 1) direct human harassment, in which humans attempt to frighten, but not kill, cormorants, 2) simulated human harassment, in which static or animated devices frighten cormorants by simulating human threats, and 3) other harassment, in which the negative stimulus is not necessarily connected to human activity.

Human Harassment

The most common form of direct human harassment is ground patrol with pyrotechnics. Patrols may occur on foot or in vehicles and may utilize a variety of pyrotechnics to frighten cormorants (and other birds). Pyrotechnics that have been discussed include various

shellcrackers, screamers, whistling projectiles, exploding projectiles, bird bangers, flash / detonation cartridges and live ammunition (Lagler 1939; Moerbeek et al. 1987; Hanebrink and Byrd 1989; Stickley and Andrews 1989; Littauer 1990a; Brugger 1995; Mott and Boyd 1995; Pitt and Conover 1996; Spencer 1996; Littauer et al. 1997; Reinhold and Sloan 1999). Live ammunition is often used because it is cheaper and more readily available than pyrotechnics (Littauer 1990a; Mott and Boyd 1995; Littauer et al. 1997). Because of potential for accidental death, those using live ammunition should also possess kill permits (Littauer 1990a; Littauer et al. 1997). Laser guns for startling birds with flashes of light have been marketed for use against DCCOs at aquaculture facilities (Desman S.A.R.L., France, Desman@wanadoo.fr); however, because their effectiveness is greatest in low light, lasers are more effective at startling DCCOs from night roosts (see: 4) Non-lethal Harassment at Night Roosts, below).

Adding labor and materials necessary to patrol a typical catfish farm with cost of pyrotechnics and live ammunition, Littauer et al. (1997) estimate harassment costs at \$132 / day. Apparently referring to Stickley et al. (1992), the authors calculate that a farm with an average of 100 DCCOs feeding at any time could experience losses of \$400 / day (replacement costs); they conclude there may be instances in which an aggressive harassment program would be cost effective. This appears to be the only attempt at a benefit:cost analysis of harassment techniques. In a survey of 281 Mississippi catfish farmers Stickley and Andrews (1989) report an average of 2.6 person-hours of harassment / day for all bird species, at an annual cost of \$7400; harassment during the 6 month period when cormorants were present averaged \$26 per day, or approximately \$4700 for the entire six months.

Surveys of aquaculturists reveal that harassment patrols are commonly utilized, despite the fact that few consider them very effective. Parkhurst et al. (1987) found that human patrols were considered highly successful or better by 20 of 66 hatchery managers (30 %), of limited success by 40 (60 %) and not successful by six (9 %). Stickley and Andrews (1989) reported that, of 244 survey respondents who reported using harassment techniques, 147 (60 %) used vehicle patrols combined with shooting to repel birds; of these 147, 13 % found this combination to be very effective, 47 % somewhat effective and 40 % not effective. Of these same 244 respondents, 21 (9 %) utilized pyrotechnics regularly; of these 21 individuals, 24 % found pyrotechnics to be very effective, 57 % somewhat effective and 19 % not effective. In both studies above, effectiveness was not defined. Brugger (1995) reported that in Florida human and dog patrols frightened birds only briefly. Spencer (1996) reported that two state fish hatcheries in Georgia used whistlers and cracker shells, and both techniques were considered partially effective against birds in general at each facility. Neither technique was considered effective against “resident” DCCOs, however. A survey by Wywialowski (1998, 1999) found that 57 % of catfish farmers surveyed shot at cormorants with intent to both kill and harass, and 55 % used vehicle patrols of some sort. Moerbeek et al. (1978) reported that flash / detonation cartridges were generally ineffective against Great Cormorants, with some cormorants leaving the facility, some moving to other ponds within the facility and others unaffected.

Cormorants are often frightened from aquaculture ponds simply because of presence of humans, and this behavior potentially may be exploited to reduce cormorant damage. Increasing overall human activity levels, keeping most valuable stocks near buildings and activity centers, and promoting visitors may decrease predation on aquaculture stocks (Hanebrink and Byrd 1989; Hodges 1989; Spencer 1996; and Reinhold and Sloan 1999). Hodges (1989) concludes that the presence of humans at aquaculture facilities during critical periods may be the most effective

means of keeping DCCOs off ponds. There do not appear to have been any analyses of this technique, however.

Various forms of aircraft have been tested for their ability to frighten cormorants from aquaculture facilities. Overall, success rates appear to be high compared to other harassment techniques, but the costs associated with this technique have also been considered high. Model aircraft have been discussed (Mott and Boyd 1995; Littauer 1990a; Conniff 1991). Littauer (1990a) states that one person operating remote controlled aircraft could effectively protect 200-300 contiguous acres (81 – 121 ha) of ponds, but compared to ground patrols with unskilled labor, this technique was more expensive. Conniff (1991) reported that model aircraft were effective, but expensive in terms of labor and equipment and that in collisions between cormorants and model aircraft, cormorants usually fared better. Moerbeek et al. (1987) tested helicopters and ultralight aircraft at an aquaculture facility in the Netherlands and concluded that the helicopter was impressive in technical possibilities, but its effects were short lived (i.e., the cormorants returned quickly); the ultralight seemed to produce better results, but few data were available. They also noted that ultralights were limited to low wind conditions.

Simulated Human Harassment

To reduce labor costs for harassment patrols, various devices, both static and animated, have been developed to simulate the threat of human activity near aquaculture ponds. These devices range from simple wood cutout scarecrows to elaborate contraptions that create startling movements, emit numerous noises and flash lights.

Human effigies / scarecrows have long been used against avian predators at many different types of agriculture fields, despite their general lack of success at preventing depredations (Lagler 1939; Inglis 1980). Increasing both realism and level of animation in scarecrows may improve their ability to scare birds, and combining scarecrows with automated sound devices (see below) may enhance the frightening effect (Littauer 1990a; Littauer et al. 1997). Improving the level of animation can be as simple as adding arms that swing in the breeze or as complex as using pop-up or inflatable scarecrows.

Stickley et al. (1995) and Stickley and King (1995) tested an inflatable effigy called Scarey (sic) Man © on catfish farms in Mississippi. This device deploys at variable intervals, popping up from the ground and bobbing around until deflating a short time later. Deployment is accompanied by a shrieking sound. For relatively short lengths of time (10 – 19 days), the device significantly reduced the number of DCCOs on the ponds (71 – 99 % reduction in number of DCCOs flushed from ponds during ground patrols); compared to replacement cost of catfish consumed (based on mean DCCO consumption rate in Stickley et al. 1992), Scarey Man devices were considered to be economically efficient (Stickley et al. 1995). However, evidence of habituation was reported, especially when day roosts were in view of the Scarey Man devices. The longer-term (46 day) study by Stickley and King (1995) considered various enhancement techniques (e.g., adding hats and masks to the Scarey Man devices, replacing them on occasion with real humans shooting guns, adding propane cannons) to decrease the rate of habituation. They found that the devices could be effective for long periods of time, especially if moved frequently; they did not find evidence that the propane cannons helped, however. The authors suggest that the device be used where DCCO depredations are “serious”.

Another animated scarecrow was described by Conniff (1991). This device was described as a jack-in-the-box scarecrow with inflatable plastic arms, revolving strobe lights, and amplified

sounds (130 dB) of horns honking, people shouting, shotguns firing and birds screaming. The device was declared “ineffectual”, indicating that “a cormorant can get used to almost anything.”

Unattended vehicles such as Trucks, ATVs and boats have been tried and recommended as means to simulate human threats (Stickley and Andrews 1989; Reinhold and Sloan 1999). There are few data on the effectiveness of this technique. Stickley and Andrews (1989) reported that one survey respondent used a parked vehicle with a scarecrow and found it very effective.

To simulate the threat of gunfire, various noisemakers have been developed. One is the rope firecracker, which sets off explosions as the rope burns (Littauer 1990a). More commonly discussed are propane / butane / acetylene cannons (Lagler 1939; Moerbeek et al. 1987; Hanebrink and Byrd 1989; Stickley and Andrews 1989; Littauer 1990a; Conniff 1991; Brugger 1995; Mott and Boyd 1995; Stickley and King 1995; Pitt and Conover 1996; Spencer 1996; Littauer et al. 1997; Simmonds et al. 1997; Reinhold and Sloan 1999). Some cannons can be programmed to go off at varying intervals and variable numbers of times. The reported effectiveness of cannons is itself variable. Stickley and Andrews (1989) reported that, of 244 survey respondents, 97 (40 %) used propane cannons. Of these, 60 % found them to be either somewhat or very effective against DCCOs (the same approval rating as for ground patrols); however, ground patrols were found to be slightly more effective (13 % of those using ground patrols found them to be very effective, whereas only 9 % of those using cannons did). Brugger (1995) reported initial success with cannons, but with relatively quick habituation. Conniff (1991) reported that butane cannons eventually became perches for DCCOs. Individuals using propane cannons in Oklahoma and Georgia reported them to be ineffective (Spencer 1996; Simmonds et al. 1997). Moerbeek et al. (1987) found gas cannons generally ineffective against Great Cormorants in the Netherlands. Littauer (1990a,b) suggests that the cannons be moved every 1 – 3 days to limit habituation, and that their use be stopped when the number of habituated birds increases, otherwise future effectiveness of the devices might be compromised.

Perhaps more importantly, use of cannons in conjunction with other harassment techniques improves their effectiveness (Stickley and Andrews 1989; Littauer 1990a; Littauer et al. 1997; Reinhold and Sloan 1999).

Other Harassment

Other means to startle birds into flight have been developed, and many have been used against DCCOs. These include amplified DCCO distress calls; sirens and other electronically generated noises; tin plates, mylar balloons, reflecting tape and other reflectors; eyespot balloons and raptor silhouettes (Lagler 1939; Barlow and Bock 1984; Moerbeek et al. 1987; Parkhurst et al. 1987; Stickley and Andrews 1989; Littauer 1990a; Brugger 1995; Mott and Boyd 1995; Price and Nickum 1995; Spencer 1996; Littauer et al. 1997; Reinhold and Sloan 1999.)

Reviewers have generally found distress calls ineffective against DCCOs for long periods of time. Hanebrink and Byrd (1989) mention that, while the ADC recommended using amplified distress calls, all the calls did was move birds to different ponds. Littauer (1990a) listed distress calls among other electronically generated noises whose effectiveness was “uncertain,” but did note observations of DCCOs apparently being *attracted* to distress calls. Mott and Boyd (1995) and Reinhold and Sloan (1999) both lump distress calls with other static and automated harassment devices that, together, varied in effectiveness. Stickley and Andrews (1989) reported that of 244 survey respondents, six (2 %) used distress calls, with two reporting them to be somewhat effective and four reporting them not effective.

Because of non-uniform application of distress calls among aquaculture facilities, conclusions of ineffectiveness may be premature. Brémond (1980), Krebs (1980) and Slater (1980) discuss the use of auditory devices to frighten birds from agricultural areas, and conclude that effectiveness can be hampered if attention is not paid to bird behavior. Simply recording distress calls and playing them back, ignoring the context in which the birds hear the sound, may not be the most effective means for frightening birds. Brémond (1980) suggests analyzing distress calls to determine what parts cause the most agitation, and manipulating these parts to produce “super-normal” stimuli (i.e. unnatural stimuli that have greater effects than natural stimuli). Slater (1980) hypothesizes that the benefits of properly-applied distress calls (whether normal or super-normal) may include 1) effectiveness at lower volumes than pyrotechnics, making them less annoying to humans, 2) slower habituation rates because of prior genetic adaptation to behavioral responses and 3) less disturbance to non-target wildlife. These suggestions should be approached with some caution as they are based largely on theoretical ethological arguments; more research with DCCOs is needed in this area.

Effectiveness of audio and visual scare tactics in this category (specifically distress calls, electronically generated noises, tin plates, mylar ribbon, flash tape, flagging, helium balloons, inflatable eyespot balloons and hawk silhouette kites) have generally been found to be low when deployed by themselves or over long periods of time (Littauer 1990a; Spencer 1996; Reinhold and Sloan 1999). Barlow and Bock (1984) reported that, in Australia, noise makers and visual scare devices were not compatible with livestock, and that habituation was a problem.

Given the general lack of success with harassment techniques used separately (largely due to habituation), many investigators have concluded that, to be effective, 1) a variety of techniques must be used, 2) techniques should be applied vigorously, 3) location of static and automatic devices should be changed frequently, and 4) the combination of techniques should be altered frequently (Moerbeek et al. 1987; Littauer 1990a; Mott and Boyd 1995; Mott and Brunson 1997; Reinhold and Sloan 1999). The concept that techniques are best applied in combination (and may in fact have a synergistic effect) appears to be shared with aquaculturists as well (Sticklely and Andrews 1989). The recommendation by Littauer (1990a) that use of gas cannons be stopped once habituation begins to occur (to prevent a decrease in their utility at a later date) could probably be applied to all forms of non-lethal harassment. Again, whether an intense program of harassment could be economically justified is unknown given the current level of knowledge of predation impacts. The ability of DCCO harassment efforts to simultaneously reduce predation by other piscivores is another important question that has yet to be investigated.

In Florida, border collies have been trained to chase waterbirds from fish-rearing facilities (J. Rodgers), however we did not find any additional information on this technique.

3) Lethal Control at Aquaculture Facilities

Reinforcing Non-lethal Harassment Techniques

Recent discussions of lethal control of DCCOs by shooting at aquaculture facilities has focused on reinforcement of non-lethal harassment techniques, rather than population reduction (Slater 1980; Littauer 1990a; Mott and Boyd 1995; Spencer 1996; Hatch and Weseloh 1999; Reinhold and Sloan 1999), including the Depredation Order (Trapp 1998). However, in some cases lethal control has also been recommended to remove problem individuals that appear immune to non-lethal harassment (Keith 1995, Pitt and Conover 1996, Spencer 1996). The observation that lethal control may help prevent habituation to non-lethal harassment techniques

is consistent with the ethological concept of psychological extinction in which individuals may, over time, fail to respond to artificial stimuli (non-lethal harassment, in this case) if the “punishment” (shooting to kill) is removed (Slater 1980). Few empirical data supporting the effectiveness of lethal control of DCCOs exist, however, and no economic analyses comparing costs of lethal and non-lethal controls with benefits of avoided predation are available. Hess (1994, as cited in Reinhold and Sloan 1999) observed that fewer DCCOs appeared to enter treatment areas where lethal control was practiced, suggesting that the control was effective to some extent. Reviewing European studies, McKay et al. (1999) found mixed opinions on effectiveness of shooting Great Cormorants to reinforce harassment efforts.

Shooting at DCCOs may not be an effective harassment technique by itself and may simply move birds from one pond to another within the same facility (Hanebrink and Byrd 1989; Simmonds et al. 1997). Simmonds et al. (1997) conclude that lethal control is controversial and often counter-productive; they do not explain in what ways it may be counter-productive, however.

Prior to March 1998, anyone wishing to practice lethal control of DCCOs at aquaculture facilities in the U.S. was required to obtain an individual depredation permit from the USFWS (as well as state permits, where applicable). Applicants were required to provide evidence that non-lethal techniques had been tried and found ineffective, and to estimate economic losses if DCCOs were not controlled with lethal means (Trapp et al. 1995). To reduce paperwork and speed up response rate for aquaculturists seeking permission to shoot cormorants, USFWS issued the DCCO Depredation Order on 4 March 1998 (Trapp 1998). The goal of the Depredation Order was not to effect large scale population reduction through lethal control. In fact, the Order predicted that the expected increase in mortality would not have a negative impact on DCCO populations. Rather, the Order was intended to reinforce and supplement non-lethal control efforts (Trapp 1998). Although the Service continues to support development of non-lethal alternative technologies for reducing cormorant impacts, it concludes that lethal control “will always have to be considered as a viable option in an effective, integrated strategy for minimizing the deleterious effects of cormorants on aquaculture” (Trapp 1998:10557-8).

Using available survey data, it is difficult to determine how frequently lethal control is used among aquaculturists. Wywiałowski (1998, 1999) reports that 57 % of respondents shot at DCCOs. Stickley and Andrews (1989) reported that 60 % of respondents (147 of 244 surveyed) patrolled their ponds, shooting at DCCOs. In neither survey was shooting to harass and shooting to kill differentiated.

Reducing Local Populations

For lethal control at aquaculture facilities to effectively reduce local DCCO populations and predation levels, the augmented mortality rate must be higher than the immigration rate. This level of lethal control may be difficult to achieve. In Bavaria, Germany, the number of Great Cormorants reported shot during the winters of 1995/96 to 1998/99 was equal to between 50 and 100 % of the average Bavarian winter cormorant population. Despite this high level of mortality, mean winter numbers of cormorants did not substantially decrease. Shooting did not remove birds from water bodies that were supposed to be protected, but simply killed migrant birds which were rapidly replaced by newly arriving individuals. Because mean GRCO winter numbers did not decrease substantially, shooting was considered an inappropriate management tool for reducing overall fish depredation in Bavaria. No conclusions were drawn concerning use of shooting to reinforce local harassment efforts, however (Keller et al. 1998; Keller 1999b). In

the Mississippi Delta, numbers of DCCOs increase over the course of the winter, reaching peaks in February and March (J. Glahn, pers. comm.). King (1996) found that three of 25 cormorants fitted with radio transmitters in the Delta flew to the Gulf Coast approximately 350 km away (one of these making the trip in two days), and concluded that “wintering birds may wander extensively and rapidly travel great distances.” Thus, in this region, similar outcomes of lethal control efforts to those in Bavaria may occur due to winter movement patterns of cormorants through and within the Delta.

Additionally, anecdotal evidence indicates that DCCOs are wary birds and difficult to kill at aquaculture facilities (Hanebrink and Byrd 1989; Conniff 1991; Price and Nickum 1995). Some empirical evidence on the difficulty of killing cormorants at aquaculture facilities appears in Hess (1994, cited in Reinhold and Sloan 1999). During 3000 person-hours of effort in which maximum take was not limited, only 290 DCCOs were killed at two aquaculture facilities in the Mississippi Delta. The Depredation Order (Trapp 1998), also citing Hess (1994), reports that investigators were able to kill only 12 % of the number of DCCOs authorized (2500) over a 19-week period. These observations are important for evaluating the feasibility of population reduction by shooting birds during the winter. Some have suggested that shooting DCCOs at aquaculture facilities may not be an effective population control technique by itself (Thompson et al. 1995; Simmonds et al. 1997).

Accidental Killing of Other Species

The issue of by-kill has also arisen within discussion of lethal control options for DCCOs, as has the problem of abuses of lethal control permits. For example, Conniff (1991) reports one aquaculture facility in California was convicted of taking 15,000 birds over five years when it had only a 50-bird permit; minimum-wage labor “ill equipped to distinguish between a water turkey [Anhinga] and a Bald Eagle” had killed not only DCCOs and herons, but also hawks and ducks that do not eat fish. Simmonds et al. (1997) reported that in Oklahoma, Anhinga, Cattle Egrets and Little Blue Herons were shot during DCCO lethal control efforts. On Prince Edward Island, legal hunting of DCCOs was found to have little impact on that species’ numbers; however, inadvertent killing of the less common and protected GRCO resulted in a 50 % decrease in the population; as a result, the PEI government began working to eliminate the DCCO hunting season (Korfanty et al. 1997). In addition to active hunting of cormorants, passive lethal control techniques (e.g. leg-hold traps) have been deemed unacceptable because of great potential for by-kills (Barlow and Bock 1984).

The DCCO Depredation Order (Trapp 1998) does not consider by-kill a significant problem, stating that 1) few threatened or endangered species are expected to be confused with DCCOs and 2) mortality to other cormorant species and Anhinga was not expected to increase significantly. [However, see PART II. POPULATION DATA AND TRENDS, Louisiana summary, for potential impacts to Neotropical Cormorants; the Anhinga is declining in parts of its range (W. Howe pers. comm.)]. Predictions of impacts to non-target species do not appear to have been based on calculations from empirical data or modeling. The Order does not call for monitoring of the impacts of by-kill beyond tallying self-reported take. The USFWS plans to decrease the level of by-kill resulting from the depredation order by 1) restricting shooting to daylight hours and 2) working with USDA/WS and NGOs to develop educational materials.

4) Non-lethal Harassment at Night Roosts

Harassing DCCOs at night roosts with pyrotechnics and other techniques until the cormorants leave has been suggested as a means of discouraging DCCOs from a geographical area on a scale larger than the typical aquaculture facility. The goal of this approach is to keep DCCOs from roosting in the area at night and subsequently to decrease the number of depredating cormorants within the harassment area during the day. Most discussion of this technique has focused on the Mississippi Delta region where it has been practiced since 1992 (Mott et al. 1998). A survey of catfish producers in southern U.S. (Wywiałowski 1998, 1999) revealed that 14 % reported using this technique. Most were in Mississippi (32 % of all Mississippi catfish producers), whereas only 6 % of producers outside Mississippi reported using this technique.

Mott et al. (1998) and Reinhold and Sloan (1999) provide the most recent summaries of the Mississippi Delta program, and are excellent resources; we briefly summarize their conclusions here. In a cooperative effort among aquaculturists, hunters, state and federal agencies, and other “concerned individuals” organized by USDA/WS, the Mississippi Delta is monitored with ground and aerial surveys, and cormorant roosts located during these surveys are harassed. Harassment occurs from two hours before sunset to half an hour after, and is repeated nightly until ≥ 90 % of cormorants have abandoned the roost, typically within 1 – 2 days. USDA/WS organizes simultaneous harassment of all known night roosts within the Delta once per month, and individuals are encouraged to practice their own harassment efforts at other times. Harassment includes pyrotechnics such as screamer sirens, bird bombs and shell crackers.

Laser guns have been marketed to startle DCCOs in low-light conditions (Desman S.A.R.L., France, Desman@wanadoo.fr, Hatch and Weseloh 1999). McKay (1999) reports that laser guns have been used effectively against GRCOs in England, Wales, France and Italy; this technique effectively reduced numbers of birds at night roosts as well as numbers feeding in nearby ponds in England and Wales. These investigators recommended this technique be utilized especially in regions where pyrotechnics will disturb other species.

Assessments of the effectiveness of the Mississippi Delta program are limited, but tend to suggest that the program may be numerically and economically effective. Surveys within harassment areas during the winter of 1993/94 counted 70 % fewer birds than the previous winter (1992/93) when there was no harassment; surveys from 1994/95 showed a 71 % decrease from number of birds detected in 1992/93 (Mott et al. 1998). Survey data from aquaculturists in 1994 revealed that 62 % within the harassment zone reported fewer problems with DCCOs than in previous years, whereas 38 % outside the zone had the same sentiments; data from 1995 revealed little change, with 74 % of aquaculturists within the harassment zone reporting fewer problems and 38 % outside reporting the same (Mott et al. 1998). These perceptions are reflected in the amount of money spent on harassment at individual aquaculture facilities: within the harassment zone, aquaculturists reported an average \$1406 decrease in expenses for harassment at their facilities in 1994, and \$3,217 in 1995. (Whether some of this decrease was due to reallocation of resources to night roost harassment, in addition to changing perceptions of DCCO impacts is not reported.) Cost of the program was \$16,757 in 1994 and \$32,302 in 1995. If these costs were divided equally among participating aquaculturists, each would have paid \$419 in 1994 and \$557 in 1995. Based on a comparison between cost of the night roost harassment program and reductions in harassment expenditures at individual aquaculture facilities, the control program was considered economically effective.

Outside harassment zones, expenditures for harassment at individual aquaculture facilities increased on average \$845 in 1994 and \$741 in 1995, suggesting that the harassment program moved DCCOs out of the harassment zone and into surrounding areas (Mott et al. 1998). King (1996) attempted to analyze the effect of the Delta harassment program on DCCOs using radio telemetry; however, the data are difficult to interpret with respect to the control program. Overall, night roost harassment was not considered to be sufficient for eliminating the need to practice other forms of DCCO control at aquaculture facilities, though it may be effective at decreasing losses (Littauer 1997; Mott et al. 1998; Reinhold and Sloan 1999).

Jackson and Jackson (1995) posed two questions about potential counterproductive impacts of night roost harassment based on bioenergetics: 1) might harassment cause *more* depredations because harassed cormorants expend more energy, and therefore must eat more, and 2) might the cormorants stay longer in the Delta, consuming more catfish, because it is more difficult for them to build up reserves for their northward migration? There are no energetics data to answer these questions; however a study (Wendeln and Becker 1999) on nocturnal disturbance of Common Terns reported that increased energy expenditure was compensated by increased foraging. The apparent success of the Delta control program suggests that these issues may not be important on a local (within the control region) scale. Also, if harassment shifts some of cormorant predation efforts to natural feeding areas such as the Mississippi River, then any increases in predation caused by harassment will be economically inconsequential. Preliminary telemetry studies suggest this may be the case (J. Glahn, pers. comm) Questions that remain about the Delta night roost harassment program include analysis of net economic impact. How do costs of the program compare with predation avoided as a result of the program? Do perceptions of decreased impact, as measured by changes in expenditures for harassment at individual aquaculture facilities, provide accurate indices of changes in DCCO impacts on net profits?

Some investigators have begun to question the long-term effectiveness of night roost harassment (Glahn et al. 2000). The number of known night roosts within the Delta has increased recently, making monitoring and harassment more difficult. Whether this increase in number of night roosts is due to harassment, to increasing numbers of DCCOs wintering in the Delta, or to a combination of factors, is unknown.

5) Provision of Alternative Prey and Foraging Sites

Provision of Alternative Prey

Fish species other than those intended for harvest may comprise a significant proportion of biomass within aquaculture ponds. These “trash fishes” may be preyed upon by DCCOs preferentially to stocked species, and some investigators have suggested active management of these species in aquaculture ponds to decrease predation on fish destined for harvest (Barlow and Bock 1994; Glahn et al. 1995; Nisbet 1995; Simmonds et al. 1997). Most discussion of this topic has focused on gizzard shad in catfish ponds (Hodges 1989; Sticklely et al. 1992; Glahn et al. 1995; Glahn et al. 1998), although it has been observed that green sunfish may have a similar effect on predation in catfish ponds (Hodges 1989; Glahn et al. 1995). The observation by Sticklely et al. (1992) that average handling time was lower for shad than for catfish (avg. = 7.2 s and 47 s, respectively) suggests one reason for the preference. Glahn et al. (1995) also suggest that schooling behavior of shad may make them more attractive prey for DCCOs.

Glahn et al. (1995) recommended stocking shad in catfish ponds in the spring to offset the increase in the percentage of catfish in DCCO diets observed during this time of year.

However, preliminary results from an experimental study of DCCO predation on catfish suggest that alternative prey (shiners, rather than shad, in this case) may not completely buffer the impacts of DCCO predation under certain conditions (J. Glahn, pers. comm.). Problems with this technique may include: attracting more DCCOs to ponds, increasing disease and oxygen stress to catfish, difficulty in managing shad (which may be less tolerant to low temperatures and oxygen levels than catfish), and lower value of catfish harvests that include “trash fishes” (Hodges 1989; Glahn et al. 1995; Mott and Boyd 1995; J. Glahn, pers. comm.). There appears to have been little investigation of these concerns, however.

In Australia, mortality rates for fish in ponds with crayfish did not differ significantly between ponds with and without cormorant predation. Fish in ponds with cormorant predation and without crayfish suffered higher mortality rates. Therefore, crayfish may have provided a preferred alternative food source for cormorants, protecting fish from predation; however sample sizes were small (Barlow and Bock 1994).

Alternative Foraging Sites

Natural wetlands, restored wetlands and defunct ponds stocked with preferred prey such as shad may shift cormorant foraging away from catfish ponds (Erwin 1995; Mott and Boyd 1995; Price and Nickum 1995) and may improve non-lethal harassment efforts by providing cormorants a place to forage when driven from aquaculture facilities (Lagler 1939; Krebs 1980; Slater 1980; Mott and Boyd 1995; McKay et al. 1999). Hodges (1989) reports that DCCOs showed a statistically significant tendency to fish in large, defunct catfish ponds (although the power of the analysis was low). As with providing alternative prey within active aquaculture ponds, however, providing alternative feeding sites may increase the survival of depredating species, allowing populations to increase and put further pressures on agricultural crops (Krebs 1980).

It appears that alternative foraging sites will function less effectively at reducing predation by themselves than in conjunction with on-site exclusion and / or harassment techniques. For these ponds to be selected preferentially by foraging cormorants, one would expect that the alternative sites would have to be of higher quality than the catfish ponds themselves, a quality that may be difficult and costly to maintain.

Although some farms may benefit from the buffering effect of shad, and the maintenance of alternative feeding sites may improve the effectiveness of other control techniques, neither of these control techniques appears to have been applied consciously within catfish production regions: Wywiałowski (1999) did not report any catfish producers using alternative prey in 1996, although this technique may have been included in the category “other.”

6) Alteration of Aquaculture Practices

This is a catch-all category for the many ways aquaculturists may decrease predation losses by altering how they run their facilities. Wywiałowski (1998, 1999) found that 10 % of catfish farmers surveyed reported using techniques under this heading. Wywiałowski also reported that 7 % of respondents used techniques in the category “other”; some of these techniques may be included in this section. Much of the discussion of these techniques for cormorant control is based on theoretical arguments, and evidence of effectiveness is mainly anecdotal.

Decrease Fish Densities

Although very little is known about DCCO selection of foraging habitat, some investigators predict aquaculture ponds with high densities of fish may be more likely to be detected by DCCOs as potential feeding sites. Fish densities in catfish ponds can be as high as 150,000 / ha (Hodges 1989). By decreasing density of catfish in ponds, aquaculturists may decrease the chance of their ponds being detected and selected as feeding sites (Lagler 1939; Barlow and Bock 1984; Duffy 1995; Jackson and Jackson 1995; Mott and Boyd 1995; Simmonds et al. 1997). Obviously, decreasing stocking rates, while possibly decreasing predation, may also reduce production. However, reducing stocking rates may solve other aquaculture problems (oxygen stress, disease, parasites, etc.) common to high stocking densities, and may have multiple benefits leading to increased production (Duffy 1995). More research is needed to evaluate whether this technique is effective for managing DCCO impacts and to determine the interplay among stocking densities, fish growth rates, disease, oxygen stress, parasites, DCCO foraging site selection, and DCCO foraging rates. Factors that stress fish and limit their growth (disease, oxygen stress, parasites, etc.) may also impact predation rates because they affect fish behavior; stressed fish may be preferentially hunted because they are less capable of escaping predation (B. Vondracek, pers. comm.).

Among catfish producers the trend has been towards higher stocking densities despite the proposed problems summarized above, suggesting that among catfish farmers the benefits of higher stocking are perceived as greater than the costs. Increased stocking rates may in fact be a response to losses due to cormorants (J. Glahn, pers. comm.).

Pond Dyes

In a similar vein, it has been suggested that cormorant predation might be reduced by adding dye to the ponds to decrease the chance that fish will be detected by cormorants in flight, or to make it more difficult for cormorants to hunt once in the water (Barlow and Bock 1984; King et al. 1995; Mott and Boyd 1995; Spencer 1996; Hatch and Weseloh 1999). There appears to have been no systematic testing of this technique, but theoretical arguments and anecdotal observations suggest that this technique may be of limited use. Spencer (1996) reports that dyes have been used but found ineffective in Georgia. J. Glahn (pers. comm.) notes that catfish ponds are naturally turbid, yet cormorants have adapted well to foraging in them. Barlow and Bock (1984) observed that various cormorant species in Australia took fish from waters with Secchi disk transparency depths of less than 10cm. Van Eerden et al. (1995) state, "The habit of mass fishing in [*P. carbo*] is only a recent phenomenon, thought to have evolved in response to turbid underwater conditions with a high stock of pelagic fishes." Hatch and Weseloh (1999) suggest that tactile feeding in DCCOs may render dyes ineffective. On the other hand, experimental work with GRCOs in Israel found that prey detection was visual, and that shiny were more readily detected than dark-colored fishes (Strod et al. 1999).

Water Turbulence

Surface turbulence (e.g. by water sprayed over ponds) may reduce visibility of prey from the air, potentially influencing DCCO foraging site selection. This technique has successfully reduced predation by some avian species, but effectiveness against DCCO predation is unknown (Littauer 1990). This technique was not reported in any surveys reviewed here for DCCOs.

J. Glahn (pers. comm.) suggests that turbulence may not be effective against cormorants with this form of aquaculture because catfish are already difficult to see due to turbidity.

However, if the density of catfish in the topmost layer of the water column (that which is visible from the air) is correlated with the total number of fish in the pond, and if cormorants utilize this visual information when selecting foraging sites, then this might still be an effective technique.

Aquaculture Facility Location

Several investigators have suggested that future aquaculture developments consider potential cormorant impacts when selecting facility location (Jackson and Jackson 1995; Price and Nickum 1995; Thompson et al. 1995). From a strictly predation-centered point of view, geographic areas along major migration flyways, such as the Mississippi Delta, are poor areas to locate aquaculture facilities because of the numbers of DCCOs and other fish-eating birds that pass through the area during migration. As early as the 1930s Lagler (1939:174) recommended “locating fish-cultural establishments outside the usual limits of the major flyways of fish-eating birds and beyond the flight range of large nesting colonies” to avoid severe predation. Because large aggregations of aquaculture facilities may be more readily detected by DCCOs than isolated ones, it has also been suggested that new facilities be located away from preexisting concentrations of aquaculture facilities (Price and Nickum 1995). Apparently this preventive strategy has not been frequently utilized, as Thompson et al. (1995:185) state, “Siting decisions often do not appear to consider all the benefit:cost factors, such as the potential for predation.”

In the past, when cormorant numbers were low, location of aquaculture sites was based largely on physical features (e.g., proper soil types, availability of water) and social features (e.g., proximity to processing plants and markets). Today, in the face of increased predation by growing cormorant populations, it seems advisable to consider the economic costs of predation within the site selection process.

Improved Facility Design

Retrofitting existing aquaculture facilities to incorporate control techniques such as exclusion has been dismissed as economically unjustifiable (Price and Nickum 1995). However, recommendations that facilities be designed to incorporate improved management techniques have been around for decades (Lagler 1939), presumably unheeded (Jackson and Jackson 1995). Mott and Boyd (1995), Price and Nickum (1995) and Pitt and Conover (1996) all suggest that aquaculture ponds be designed to make netting and wiring more economically efficient. Simmonds et al. (1997) recommend smaller ponds because larger ones have higher predation rates, perhaps because they are easier to detect from the air. These authors also recommend maintaining trees around ponds not only to decrease their delectability from the air, but also to make taking off and landing difficult for DCCOs. The trend of increasing pond sizes suggests that the costs of cormorant predation and control continue to be overlooked when decisions are made of the most “economically efficient” pond size.

Alter Stocking Practices

Because young fish may be more vulnerable to DCCO predation, biologists have suggested that aquaculturists in DCCO wintering areas delay stocking of young fish until later in the spring, when DCCOs have returned to their breeding territories (Moerbeek et al. 1987; Brugger 1995; Glahn and Stickley 1995; Glahn et al. 1995; Mott and Boyd 1995; Simmonds et al. 1997). However, J. Glahn (pers. comm.) states that “delaying stocking later in the spring is probably not a viable option because changing water temperature during spring increases the risk of stress-related mortality and disease of fish transfer.” No systematic analyses appear to have

been performed comparing the benefits of reduced DCCO predation to the costs of these other sources of mortality. Glahn et al. (1995) also suggest stocking larger fish.

Passive Harassment

Human activity may reduce DCCO numbers, and it has been suggested that daily activities at aquaculture facilities be managed to more effectively discourage predation. This can be as simple as scheduling regular activities around ponds, encouraging tourism, and stocking most vulnerable fish near areas of high human activity (Moerbeek et al. 1987; Brugger 1995; Mott and Boyd 1995; Pitt and Conover 1996; Simmonds et al. 1997). Age-class segregation of fish may also allow for more efficient harassment efforts because efforts could be focused on the ponds with the most vulnerable and / or valuable age classes. (Brugger 1995). Currently, multiple age classes commonly occur in aquaculture ponds, making application of this technique difficult (Price and Nickum 1995). There appears to have been no systematic analysis of this technique.

Alteration of Fish Behavior

Hatch and Weseloh (1999) suggest that alteration of fish behavior may reduce predation impacts, but they do not give any details about how this might be accomplished. One potential strategy may be to solve disease and oxygen stress problems (see: 8) No Control, below). McKay et al. (1999) suggest that naïve, hatchery-raised fish might be taught to better avoid predators, but mention that attempts to use this technique have been unsuccessful thus far.

Chemical Avian Repellants

Avitrol has been used on Herring Gulls in Florida, however there is no information on use at aquaculture facilities, and it may not be ideal for hatcheries (Brugger 1995).

Conditioned Taste Aversion

For individuals of some vertebrate species, feelings of illness associated with consumption of a particular food item may lead to future avoidance of that food type. This natural response may be manipulated to reduce predation on specific agricultural and aquaculture crops. Preliminary research in England found that captive GRCOs fed trout fingerlings treated with carbachol developed a strong and long-lasting aversion to trout. However, more research is needed to determine if this technique can be successfully applied to wild cormorants (McKay et al. 1999).

Manage Other Avian Species

Other species of birds may act as visual cues to DCCOs, drawing them to aquaculture facilities; managing them may result in reduced DCCO impacts (Hanebrink and Byrd 1989; Hodges 1989; Price and Nickum 1995). For example, Hanebrink and Byrd (1989) point out that non-piscivorous birds such as Cattle Egrets may attract piscivorous birds. More research is needed to determine if this would be an effective control technique. Legal protection of these other avian species is also a concern with this technique.

7) Population Control

There has been some discussion in the literature of controlling DCCO population growth to decrease predation pressures on aquaculture regions during the winter. Glahn et al. (1999:30)

conclude that currently available DCCO control techniques “are of limited effectiveness and are becoming increasingly difficult to implement.” They predict that increases in DCCO population sizes will further reduce the effectiveness of these techniques and ultimately propose management of DCCO populations that utilize the Mississippi flyway (including shooting birds at winter roosts and breeding colonies through an expansion of the Depredation Order). Reinhold and Sloan (1999) suggest that “management implications should focus more on the long term goals of managing DCCO populations,” and state that a cormorant management plan cooperatively developed by USFWS, USDA/WS and state wildlife fisheries agencies is one way of meeting this goal. Aquaculturists have also requested population control of DCCOs (Depredation Order, Trapp 1998).

By analyzing band-recovery data, Dolbeer (1991) found that birds in the southern Mississippi region originated from breeding areas as far west as Alberta and as far east as New England. The data suggest that, while lakes Michigan and Superior were the most important sources of DCCOs wintering in the lower Mississippi Valley, significant numbers of DCCOs also came from the region between Saskatchewan and Eastern Lake Ontario (Table 27). Hatch (1995:19), citing Dolbeer (1991), states that “wintering birds that eat a Mississippi farmer’s catfish could come from anywhere across the 3000 km breeding range of the populations that winter there.” Based on these data, controlling breeding DCCOs to reduce numbers of birds on the wintering grounds would require a tremendous effort to be successful (Depredation Order, Trapp 1998). No attempts have been made to estimate the costs of such a program. Because not all birds in these breeding areas utilize the same wintering areas, population level control would also impact DCCOs that are not causing depredations. Currently, satellite telemetry is being used to further investigate the migration patterns of cormorants wintering in Alabama, Arkansas, Louisiana and Mississippi. Preliminary results of this research also indicate that birds wintering in the Delta region originate from across the Interior U.S. and Canadian population’s breeding range (S. Werner pers. comm.).

Table 27. Percent of winter band recoveries (1923-1988) that occurred in the Lower Mississippi Valley for birds originally banded during the nesting season in various geographic regions (Dolbeer 1991).

Banding region	No. recovered in winter	% winter recoveries in Lower MS Valley
Alberta	149	9
Saskatchewan	120	46
Manitoba	25	40
Dakotas/Western Minnesota	134	46
Lakes Michigan and Superior	50	70
Lake Huron	93	42
Eastern Lake Ontario	16	38
New England Coast	215	11
St. Lawrence River	67	6
Gulf of St. Lawrence	16	0

Control of cormorant populations through shooting at winter roost sites in aquaculture areas has only recently been introduced (Glahn et al. 1999); there has been no discussion in the scientific literature of the effectiveness of this technique. Observations on the lack of success of

winter hunting to reduce cormorant populations in Bavaria, Germany, due to the migratory nature of this wintering population, may be applicable here (see discussion in Reducing Local Populations, above).

In discussion and modeling of management scenarios for European GRCOs, it has been suggested that reducing the size of the overall population may not result in equivalent reductions in the number of cormorants occurring at high quality foraging areas, such as aquaculture facilities (Bregnballe et al. 1997; Van Eerden and van Rijn 1997). Because foraging habitats vary in quality and cormorants are efficient at detecting high quality foraging sites, overall population reductions may first cause birds to disappear from least preferred or low quality areas, and declines may be less marked in high quality areas (Hodges 1989; T. Bregnballe, pers. comm.). Additionally, scale of damage caused by cormorants may not be directly related to total numbers in local areas (Bregnballe et al. 1997) For example, approximately 60,000 cormorants are estimated to winter in the Mississippi Delta region; however, not all individuals may feed at aquaculture facilities, and of those that do, feeding rates may vary (e.g. Glahn et al. 1995).

According to J. Glahn (pers. comm.) the increase in numbers of DCCOs wintering in the Delta has been proportionally larger than the increase in numbers of catfish ponds in the area. This is an important observation because it indicates that cormorant foraging patterns can not be explained by simply referring to concepts such as density dependence. Changes in aquaculture practices, such as increasing stocking densities or altering harassment regimes, as well as changes in cormorant populations, likely influence cormorant foraging habitat selection and therefore affect density dependent mechanisms. A better understanding of the relationship between these factors and total population size is needed if population goals are to be set.

Even large-scale reductions in DCCO population are unlikely to eliminate the need to continue local exclusion and harassment efforts. Aquaculture ponds, with high stocking rates and lack of escape cover for fish, will remain high quality foraging sites (and therefore will continue to be subject to predation by the remaining cormorants) unless exclusion and / or harassment are used to make these ponds unattractive or unavailable. Whether population control is cost effective will depend on cost of implementing population controls relative to savings in reduced on-site controls. If, for example, population controls effectively reduced the Delta cormorant population by half (back to early 1990s levels), would the avoided predation and the reduced expenditures for on-site and night roost harassment be more than that spent on population reduction efforts?

Density-dependent effects may determine whether population control will lead to population stabilization below natural carrying capacity, as was indicated through modeling of wintering GRCOs in France (Frederiksen and Lebreton 1999, Frederiksen et al. 1999). Bregnballe et al. (1997) modeled several scenarios for controlling European GRCOs (e.g. preventing establishment of new colonies, cutting nesting trees and disturbing nesting birds, reducing reproductive output, introducing hunting season, and combining these techniques). While all management techniques led to model equilibria lower than that predicted without management, prevention of colonization in new habitat was found to be the only technique that was not buffered by density dependent effects. (For further discussion of this model, see PART VI. POPULATION MODELING, Great Cormorants in Europe) Density dependence has been identified in several population parameters for the northwest European GRCO population (Van Eerden and Gregersen 1995).

8) No Control

The decision to not control DCCOs must be included on the list of management options available at aquaculture facilities, even though this option often appears to have been overlooked (Nisbet 1995). Individual aquaculturists have been encouraged to compare economic impacts of DCCO predation to costs of control and then to make rational decisions on appropriate level of control, accepting that no control may be the most appropriate option (Parkhurst et al. 1992; Thompson et al. 1995; Littauer et al. 1997; Reinhold and Sloan 1999). Some researchers have also observed that DCCO depredations are often considered economic losses while other sources of mortality (e.g. disease and oxygen stress) are considered just a cost of doing business, and they question whether the focus on cormorant predation is appropriate (Conniff 1991; Parkhurst et al. 1992; Jackson and Jackson 1995).

Analyses of DCCO control should consider cost effectiveness of cormorant control relative to preventing other sources of loss. Citing USDA data, Conniff (1991) and Price and Nickum (1995) report that, although DCCO predation is highest on the list in terms of human emotion, disease and oxygen stress are larger sources of loss to the catfish industry as a whole. Solving these problems may prove more economically efficient at reducing losses than controlling DCCOs, however there appears to be no economic analyses comparing the two options. Additionally, stressed and diseased fish may be more vulnerable to DCCO predation; resolving these problems may have a secondary benefit of reducing DCCO impacts (B. Vondracek, pers. comm.). Some investigators have even suggested that DCCO predation may benefit some farms by removing diseased fish and reducing stocking rates to more suitable levels (Jackson and Jackson 1995). Wywiałowski (1998) reports that aquaculturists who reported oxygen or disease problems were more likely to report wildlife-caused losses; the correlations were significant, but did not establish causality. Hodges (1989) also observed a strong tendency for DCCOs and other piscivores to feed in catfish ponds where fish were weakened or dying due to disease, stress or weather. In Israel, experimental work with captive GRCOs found that dead fish were chosen significantly more than live fish (Strod et al. 1999).

The USFWS has stated that more economic information is needed on the relative impacts of avian predation compared to other aquaculture losses (Trapp et al. 1995). Therefore, a more systematic approach to DCCO predation should be taken, and if DCCOs are not found to be the greatest source of aquaculture mortality, research should be redirected towards more effective means of impact abatement (Duffy 1995).

9) Conclusions: Managing Impacts to Aquaculture

Despite lack of data on DCCO impacts to aquaculture, predictions can be made on effectiveness of some techniques. Local control techniques vary in ability to reduce predation. Enclosure devices can completely eliminate DCCO predation; therefore, efforts should be continued to make this technique less costly. Depending on levels of impact, harassment may also be cost effective, and combinations of harassment techniques are more effective than any such technique used alone. Lethal control may reinforce non-lethal harassment techniques, but without sufficient education, monitoring and enforcement, this technique may lead to negative impacts to other species. Many other local control techniques may be available, but more research is needed to determine which are the most cost effective. Enclosure and other techniques would be more cost effective if aquaculture facilities were designed to better

accommodate them. Many local DCCO control techniques reduce predation impacts from all avian piscivores, an economic benefit not shared with DCCO population control.

Cost effectiveness of DCCO population control, even if predation impacts are large, is hampered by several factors: 1) control of breeding cormorants to reduce predation in main aquaculture regions would require intense, ongoing efforts, coordinated between the U.S. and Canada across an extremely large geographical area; 2) population control during the winter in aquaculture regions may be hampered by migration patterns and high mobility of DCCOs and by difficulty of shooting birds at aquaculture facilities; 3) density-dependent effects may counteract population reduction efforts; 4) relationships between overall population size and local predation rates may not be direct; even large reductions in the DCCO population may not eliminate need for continued local control; 5) benefits of population control are limited to reduced predation by cormorants only, whereas exclosures and harassment reduce predation pressures by multiple species. In a review of European GRCO control options, van Eerden and van Rijn (1997:42) discuss important components of population dynamics and conclude that “large-scale interference in breeding colonies seems unjustified for solving reported problems on the flyway. Local protection measures for fish stocks seem more effective...”

“No control” may be the best management option if net losses to DCCO predation are found to be lower than costs of control. Additionally, management techniques to reduce other sources of aquaculture losses may be more cost effective than managing DCCOs. For example, diseased and oxygen stressed fish not only grow poorly, but may also be hunted preferentially by avian piscivores; managing these sources of loss may also reduce predation impacts.

Who pays for cormorant control was not addressed in any of the studies we reviewed; nevertheless, this is an important consideration. When the catfish industry first began to grow, cormorant populations had been reduced by a number of factors, especially pesticides. After these pesticides were banned (and DCCOs became a protected species) populations increased, and the cost of control shifted to aquaculturists themselves. Population control would likely shift the cost of control to the government and ultimately to taxpayers, especially if undertaken at breeding colonies. If the government decides that such a subsidy to the catfish industry is warranted, it should seriously consider whether the money would be better spent subsidizing local control efforts. Analysis of such subsidies should consider whether techniques reduce impacts of one species (e.g. population control) or multiple species (e.g. exclusion). The Bavarian government, for example, subsidizes wire exclosure systems, but not population control (T. Keller, pers. comm.).

MANAGING IMPACTS TO VEGETATION AND HABITAT

Through normal nesting activities, DCCOs impact vegetation on a local level, affecting both plants and animals that depend on vegetation for cover and nesting, such as other colonial waterbirds. Although DCCO impacts on habitat have occurred throughout the evolutionary histories of these species, some interspecific colony dynamics may prove deleterious to some species in human-altered ecosystems where habitat is limited. Humans themselves may be impacted when DCCOs kill trees valued for aesthetics or timber (see Part IV. Impacts to Vegetation and Colonial Waterbirds). This section reviews methods for reducing these impacts.

Management may occur at two levels: 1) local control, in which specific impacts are addressed and only those cormorants involved are managed, and 2) regional / population control, in which attempts are made to hold entire DCCO populations below their natural carrying capacities, thereby limiting the potential for cormorants to cause specific damages. Within this

section, *local* will generally refer to individual colonies or islands, while *regional* will refer to geographical assemblages of colonies such as those occurring within large lakes or within individual states' boundaries.

Techniques for managing local or regional cormorant populations to prevent impacts to fisheries are similar to those used to prevent impacts to vegetation and habitat. Therefore, data from past attempts to manage DCCO populations to reduce impacts to fisheries are included here when discussing effectiveness of population management techniques at reducing DCCO numbers. Based on these combined data, effectiveness of population management at reducing impacts to vegetation and habitat are discussed. Effectiveness of population management at reducing impacts to fisheries is discussed in *Managing Impacts To Fisheries*, below.

Techniques

Harassment of Adults

Non-lethal harassment techniques such as those used to keep DCCOs away from aquaculture ponds (see: 2) Non-lethal Harassment at Aquaculture Facilities, above) have occasionally been suggested as a means of disrupting existing DCCO colonies or preventing new ones from forming. Although some conservation organizations and private individuals have called for more investigation of non-lethal harassment (see Appendix F in USFWS 1999a,b), we found no record of this technique being used by itself; when non-lethal harassment techniques have been used, they have always been accompanied by lethal techniques such as nest destruction (with the killing of eggs and young). Harassment has also been an inevitable consequence of other control techniques, and it is difficult to separate out its relative contribution to overall effectiveness.

Potential problems with harassment include impacts to non-target wildlife and lack of effect on total population size. Because both audio and visual scare devices are generally not specific for DCCOs, they may scare away individuals of other species as well. If the goal of control is to protect nesting birds from habitat alteration by DCCOs, such an effect would obviously be counterproductive. This technique may be effective at protecting plants at specific locations where other nesting birds are not of concern, but it is expected that benefits would be local, shifting DCCO impacts elsewhere.

Although harassed birds may be prevented from reproducing during a specific breeding season or suffer reduced reproductive success because of time and energy expenditures involved in finding new nesting areas, this technique is generally believed to have negligible impacts on local populations when practiced on a relatively small scale (USFWS 1999a,b). However, if prevention of colonization is practiced intensely, annually and over a large enough area, this technique may be effective at managing regional populations (Bregnballe et al. 1997).

Bixby Island, Vermont:

When DCCOs were first observed nesting in 1996, the owner of the island, assisted by USDA/WS and VDFW, removed nests with chicks and eggs (238 total nests, 1996 – 1997), effectively preventing reproduction those years (see Nest / Tree Removal, below). In 1998, 148 DCCOs were reported loafing on the island, but were displaced using pyrotechnics, propane cannons, distress calls, and removal of 15 nests (USFWS 1999a,b).

Oneida Lake, New York:

From August to September 1998, USDA/WS used pyrotechnics, nest destruction and lethal take to move DCCOs off the lake and reduce predation on sport fishes. The combined effort was considered effective (USFWS 1999a,b). Harassment was also used successfully in 1999 (R. Miller, NYSDEC, pers. comm.).

Lake Ontario, New York:

DCCOs were first observed nesting on Calf Island in 1997, and all 75 nests were destroyed that year. In 1998 non-lethal harassment and destruction of 175 nests with eggs was used. "Terror balloons" and habitat manipulation (cutting of large nesting trees) are the only non-lethal techniques mentioned, though apparently others were used. The combined techniques were considered successful at preventing colonization (USFWS 1999a,b). R. Miller (pers. comm.) notes that harassment has been used successfully on Gull and Bass islands since 1994 to prevent DCCO colonization.

Nest / Tree Removal

This technique has been used to manage DCCOs locally (to eradicate colonies) and regionally (to reduce populations). The technique has been used on both ground and tree nests, and often coincides with destruction of eggs, nestlings and adults.

Nest destruction on the ground simply involves the physical breakup of nest structures. The means by which nests can be removed from trees is rarely discussed. Bédard et al. (1999) suggest that dislodging nests with long poles could be used in the St. Lawrence Estuary, Québec control program (below), but apparently the technique has not been tried. Nest removal activities on Bixby Island (see Lake Champlain, Vermont (USFWS 1999a), below) involved two 300-gallon gas-powered pumps hooked in tandem, pumping water through fire hoses at 400 gallons / minute, 120 pounds / square inch (provided by the Grand Isle Fire Department when state equipment proved inadequate). Such pressure was said to be able to sweep a man off his feet, but barely able to dislodge a cormorant nest 50 feet up in a tree. One hundred of 112 nests were successfully removed during two days of spraying (Pyne 1996). In 1996 and 1997 combined, 238 nests were removed with this method on Bixby Island, and even though not all nests could be removed, no reproduction was documented in either year (USDA/WS 1998). Entire trees have been removed, both in private (Anderson and Hamerstrom 1967) and official control efforts (USFWS 1999a,b). However, because DCCOs frequently nest on the ground, tree removal may only be effective where the substrate is inappropriate for nesting or the threat of mammalian predation is high.

Effectiveness of nest / tree removal appears to be variable, depending on local conditions and goals of control. In many private actions the goal of population control appears not to have been very successful. Only in the Wisconsin: case studies (below) was tree removal implicated in complete colony abandonment (without evidence of re-nesting near by). For other private actions, re-nesting appeared to take place among about half of disturbed pairs; in these cases, control activities may not have eradicated colonies, but they did reduce reproduction somewhat. In the Lake Champlain, Vermont (USFWS 1999a), and Lake Ontario, New York (USFWS 1999b), case studies (below) the goal of preventing colonization on specific islands by DCCOs was achieved through these techniques.

One of the problems mentioned with nest removal, even when successful at preventing colonization attempts, is that effects cannot be considered permanent; control must be repeated each time cormorants attempt to nest in areas of concern. No control technique offers permanent

solutions, however. Another concern with nest tree removal is that it may eliminate nest substrates for other species. Where DCCOs have already made trees unsuitable for nesting by other species, this may not be an issue; however, removing nest trees may shift DCCO nesting and move them into conflict with other species.

Wisconsin:

Anderson and Hamerstrom (1967) report that DCCO nest trees were cut down in colonies on the Okee Flowage around 1919, on the Fox River prior to 1955, and on the Petenwell Flowage in the late 1950s. In all cases colonies were reported to have been permanently abandoned as a result of these activities.

The Hat Island, Green Bay, DCCO colony was heavily persecuted by fishermen who destroyed ground and tree nests and many of the birds (Matteson 1983). The colony disappeared sometime around 1956 or 1957 (Lound and Lound 1956; Anderson and Hamerstrom 1967), but had apparently been reestablished by the 1990s (Kaiman and Nelson 1995). Persecution may have caused a shift from ground to tree nesting in this colony (Kaiman and Nelson 1995).

Lake of the Woods, Ontario:

Omand (1947) reports that fishermen destroyed 127 DCCO nests in this colony (apparently all present) in June 1944. Upon return in July of the same year 150 nests were found, suggesting that the initial control effort did not have the intended effect of reducing the local population and did not prevent reproduction that year.

Lake Winnipegosis, Manitoba:

McLeod and Bondar (1953) report 943 occupied nests on Whisky Jack Island reef, 20 % containing recently hatched eggs on 20 June 1945. All eggs and young were destroyed, as apparently were nests. A second visit on 13 July of the same year revealed 495 nests had been rebuilt, each containing two or three eggs within. The authors also report that 1617 nests with eggs and / or young were completely destroyed on Cormorant Island on 24 June 1945. A second visit on 26 July of the same year revealed 643 nests had been rebuilt, each containing 1 – 2 eggs.

Blackbill Islands, Lake Huron:

At a colony off Pointe au Baril, bass fishermen destroyed 181 eggs, broke up 80 nests and shot six adults in May 1946. The colony apparently rebuilt nests on adjacent rocks using the same sticks; on 8 August 1964, 43 nests were counted, all of which appeared to have been used that summer (Baillie 1947).

Prince Edward Island:

“...over the last four years, about 4,000 – 5,000 birds have resided on Ram Island, accounting for 80 – 85 % of all cormorants on PEI. In 1996, no cormorant nests were found on this island. On the other hand, the adjacent island, which had few cormorant nests prior to 1996, was found to have 4,500 cormorants in 1996. The cormorants moved from Ram Island to the adjacent island because of persecution, harassment, and nest disturbance. Double-crested cormorants, which are resilient birds, ‘got up and left.’” (Korfanty et al.1997:28)

Egg and Nestling Destruction

DCCO eggs and nestlings have been destroyed in attempts to reduce recruitment into populations and to eliminate colonies at specific locations (in conjunction with other forms of

harassment). Few specific conclusions can be drawn of effectiveness of these techniques at reaching either goal because most available data are anecdotal. In general, egg and nestling destruction may not be effective at completely eliminating reproduction for individual DCCOs during a nesting season because of their tendency to renest after disturbance. Effectiveness at regulating populations with this technique are inconclusive. Ability of this technique to remove colonies appears variable; newly established colonies appear to be more easily eliminated than well established ones.

When specifically discussed, techniques for destroying eggs and nestlings have included crushing with rocks, feet or other objects (Price and Weseloh 1986; Ewins and Weseloh 1994; Sheppard 1994/5), as well as throwing both eggs and nestlings into the water (Sheppard 1994/5). Older young have been clubbed, strangled and shot (Hobson et al. 1989; Sheppard 1994/5). In 1999, the Vermont Department of Fisheries and Wildlife requested permission from USFWS to asphyxiate by CO₂ all chicks on Young island that survived oiling efforts (see Lake Champlain, Vermont (USFWS 1999a), below). USFWS denied this request, citing a lack of data to show that survival of these young would hinder the overall control effort (USFWS 1999a). These techniques appear effective at killing eggs and nestlings, however costs appear not to have been analyzed.

Observations suggest that when renesting occurs within the same nesting season, colony size (and presumably reproductive output) may be reduced by about half (see Nest / Tree Removal, above). For Lake Winnipegosis in general, McLeod and Bondar (1953) concluded that destroying the first eggs or young of the season resulted in about 50 % of reproducing pairs renesting immediately. However, exceptions occur (see Lake of the Woods, Ontario:, above).

Birds that do renest may also experience reduced reproductive success, but this effect appears not to have been investigated. However, Ludwig and Summer (1995) do report one instance in which egg destruction had little noticeable impact on clutch sizes and fledging rates. Factors that determine which individuals or colonies renest appear not to have been investigated, but timing may play an important role: renesting is generally more likely to occur during the beginning of the nesting season than at the end.

Some cases are known where local DCCO populations continued to rise despite egg and nestling destruction efforts. Hobson et al. (1989) report increasing DCCO populations on Lake Winnipegosis, Manitoba, despite destruction of eggs, nestlings and adults by local fishermen. Number of nesting DCCOs on Lake of the Woods (both Minnesota and Ontario sides) increased from an estimated 50 pairs in 1929 to at least 472 pairs in 1939 despite heavy persecution by local fishermen (Baillie 1939, 1947). DesGranges and Reed (1981) report that the DCCO colony on Ile aux Pommés, St. Lawrence Estuary, had grown considerably since 1963, and at a faster rate than the St. Lawrence Estuary population as a whole, despite control efforts 1978 – 1980, in which all eggs and young were destroyed, and almost annual disturbance by researchers occurred (which resulted in increased egg and chick predation by gulls). The authors also report that on Iles Rasades local residents destroyed many clutches several times over the previous 20 years, but the colony had quadrupled in size during the last 10 of these 20 years.

In contrast, some investigators have suggested that egg and nestling destruction on a large scale (often in conjunction with killing adults) may have slowed the growth of populations (Weseloh and Collier 1995), stabilized populations or contributed to declines (Ewins and Weseloh 1994; Sheppard 1994/5; USFWS 1999b). However, in some regions such as the Great Lakes effects of persecution are difficult to separate from those of environmental contaminants (e.g., Ewins and Weseloh 1994). McLeod and Bondar (1954) concluded that consistent

destruction of eggs and young appeared to reduce the breeding population fairly effectively on Lake Winnipegosis in the 1940s and 1950s (see Lake Winnipegosis, Manitoba, below). Chapdelaine and Bédard (1995) state that there is no doubt illegal control on the Great Lakes (which has included egg and nestling destruction) prevented some colonies from expanding. Anderson and Hamerstrom (1967) discuss the Lac Du Bay colony in Wisconsin which persisted (though apparently undergoing a decline) despite high levels of persecution 1956 – 1963, the highest eggshell pesticide levels of all colonies tested in the state (48.9ppm DDT+DDE+TDE, mostly DDE*), and considerable paper mill effluent so bad that “the water has become scummy and darkened, sulfite wastes abundant, and poor fishing has discouraged the fisherman.”

Effectiveness at causing colony abandonment appears to vary. Most recently, egg and nestling destruction, in conjunction with nest destruction and sometimes harassment, has been found to be effective at preventing re-nesting for a year or more at newly established colonies (USFWS 1999a,b). Price and Weseloh (1986), citing Quilliam (1973), state that unspecified activities by fishermen near Kingston, Ontario probably caused DCCOs to abandon colonies on Snake and Salmon islands. Weseloh and Collier (1995), on the other hand, describe egg destruction efforts on Georgian Bay, Lake Huron, as unsuccessful because birds relayed, often on different islands. Baillie (1947) reports that on the Mink Islands, Lake Huron, 150 adult and young cormorants were shot at a colony with 40 nests in 1945; in 1946 the colony was still active and had grown to 50 nests. Baillie (1947) also discusses the DCCO colony on Mouse Island, Lake Huron. Of the three colonies in the North Channel, Mouse Island received the most attention from fishermen who made special trips there to destroy eggs and young. The colony persisted despite their efforts, leading them to register an official complaint to the Ontario Department of Lands and Forests requesting that the government control the colony. The request was denied. Finally, on an island off the coast of Maine, one colony was visited seven times during 1968, with cormorant eggs and young destroyed on each visit. Attempts to eradicate the colony continued annually until 1972, apparently without success (Krohn et al. 1995).

There appear to be few technical problems with accomplishing the goal of killing eggs and nestlings. Other than asphyxiation with CO₂, equipment needs are minimal. However, all techniques except shooting are limited to ground nests because climbing trees is not only time consuming, but it can be hazardous in trees dying from the effects of cormorant activities. One solution proposed to bypass the latter problem was to use trained monkeys to destroy eggs in Great Cormorant tree nests, but this technique does not appear to have been used (Van Dam 1997).

Given a lack of systematic investigations of these techniques, it is not possible to assess the economic effectiveness of these techniques at reducing cormorant impacts.

Egg Oiling / Spraying

Various substances have been applied to cormorant eggs to kill embryos. White mineral oil (variously: Daedol 50, Daedol 50 NF and DAEDOL 55 USP®) has been used most recently (Blokpoel and Hamilton 1989, Christens and Blokpoel 1991, Shonk 1998, Bédard et al. 1999) because of its reported lower toxicity and fewer environmental impacts compared to other substances that have been used, such as oil/water/formalin (Gross 1951) or acetic

* Ludwig and Summer (1995) state that the Great Lakes DCCO population in general could not reproduce at replacement rates until egg DDE levels fell to <10ppm, and population growth could not be sustained until levels dropped to 6-8ppm.

acid/formalin/cresylic acid/glycerine/water (McLeod and Bondar 1953). Other substances applied to eggs include “Gelfa” (an alcohol-soluble plastic solution), “Krylon” (a plastic aerosol) and shellac (DesGranges and Reed 1981). Although some of these substances may kill embryos through toxic effects, most are expected to prevent the exchange of gasses through the shell, causing asphyxiation (Blokpoel and Hamilton 1989; Christens and Blokpoel 1991; Shonk 1998). All citations above refer to control of DCCOs except Blokpoel and Hamilton (1989) who tested effects of white oil on chicken and Ring-billed Gull eggs.

Effectiveness at killing embryos is high, with mortality rates approaching 100 % when the oil / solution is applied correctly (Gross 1951; DesGranges and Reed 1981; Blokpoel and Hamilton 1989; Christens and Blokpoel 1991; Shonk 1998; NYSDEC 1999; Bédard et al. 1999). Although laboratory tests found oiling ineffective when applied only to part of an egg (Blokpoel and Hamilton 1989), field tests in which only the tops of eggs were sprayed were highly successful, indicating that careful application to the entire egg surface may not be necessary (Christens and Blokpoel 1991; Bédard et al. 1999). Egg-rolling activities by parents may assist in covering the entire surface (Christens and Blokpoel 1991).

Egg oiling / spraying has essentially the same result as that of killing eggs and nestlings, but with the added benefit that cormorants are less likely to abandon nests and lay replacement clutches, making the technique more effective as an annual treatment. Bédard et al. (1995a:81) report, “None of the eggs in 427 experimentally treated nests hatched in 1987 and nearly half were still tended by adults 49 to 59 days after laying (the remainder having been abandoned and scavenged).” Shonk (1998) reports, “Of the eggs treated with oil, 49 % were incubated past the expected hatching date. The remainder of the eggs were lost during the incubation.” Human disturbance during oiling increases predation by gulls (Shonk 1998), which may cause some cormorants to renest. However, no data are available on the rate of renesting after oiling, either in the same colony or elsewhere.

Effectiveness at population control through egg oiling is more uncertain. Because DCCO egg laying is not synchronous, only one spraying / summer may not treat enough of the eggs laid that nesting season to have a significant impact on the population, and multiple oiling efforts to overcome this problem may increase the cost of control beyond acceptable levels. Both problems have been cited as reasons for the failure of the Maine: case study (below) by Dow (1953). Modeling of the St. Lawrence Estuary, Québec (below and PART VI. POPULATION MODELING) suggested that egg oiling alone would not be sufficient to bring DCCO numbers to target levels within the desired time frame, leading to the selection of a combined strategy of egg oiling and shooting adults (Bédard et al. 1995a); however, effectiveness of the combined techniques was greater than predicted by the model (Bédard et al. 1999). Additionally, most tree nests are inaccessible to spraying by conventional techniques, limiting control to ground nests (Bédard et al. 1995a). Even for ground nests, meeting oiling goals may be logistically difficult: Bédard et al. (1999) report the St. Lawrence Estuary, Québec control program goal was 7000 eggs oiled / year, but the average number oiled was 5019. Lack of inter-colony migration rates also makes it difficult to predict the minimum scale of control necessary to overcome effects of immigration.

Christens and Blokpoel (1991) also mention egg shaking as a technique that may kill embryos without destroying the egg, but conclude that egg oiling is just as effective and less labor intensive. Hobson et al. (1989) and Sheppard (1994/5) also mention that fishermen have used flame throwers to kill eggs without breaking them, apparently with success.

Maine:

From 1944 to 1954 USFWS sprayed over 180,000 DCCO eggs with oil and formalin to kill embryos (Hatch et al. 1995) and “to reduce [DCCO] numbers until they no longer prove to be a serious menace to the weir fishermen” (Gross 1944:536). According to Gross (1951:536), “We confidently predict that in a relatively short time the number of gulls and cormorants will be reduced by the control measures to a point where they will no longer be an economic problem.” Not everyone agreed that economic impact had been demonstrated, however: Dow (1953:1) stated, “It should be noted that there is also no definite evidence indicating that these populations should be controlled; or, if control is needed, to what extent it is desirable.” Despite the fact that the goal of the program was to reduce the economic impact of DCCOs to fisheries, economic impacts do not appear to have been monitored. Rather, effectiveness of the control program was measured by number of eggs sprayed (as an index of population size) and by how many of ten monitored colonies were abandoned. Because there was no clear downward trend in number of eggs sprayed, and only one of these colonies was abandoned during treatment years, the control program was declared ineffective and oiling ceased in 1954 (Hatch et al. 1995).

Prior to this control program, New England DCCO populations experienced a rapid increase in size (Drury 1973). While the control program did not reduce population size as hoped, it may have helped slow the growth rate. Gross (1944) conservatively estimated 20,000 individual DCCOs nesting in Maine in 1943; Dow (1953) estimated a minimum 22,470 adults in Maine in 1952. DCCO breeding populations subsequently grew slowly, if at all (Drury 1973). Recent population estimates for Maine are around 17,000 to 18,000 nesting pairs (see PART II. POPULATION DATA AND TRENDS, Maine).

Avian Contraceptives

In 1999 USFWS received a request from VDFW to administer avian contraceptives DiazaCon and Nicarbazin to DCCOs in addition to other control activities on Young Island (see Lake Champlain, Vermont (USFWS 1999a), below). Results were expected to be similar to egg oiling. Permission was not granted to VDFW because neither contraceptive was registered for use on DCCOs (USFWS 1999a).

Killing Adults

Private DCCO control efforts have often included shooting adults in addition to destroying eggs and young (Matteson 1983; Hobson et al. 1989; Ewins and Weseloh 1994; Sheppard 1994/5; Carter et al. 1995; Jackson and Jackson 1995; Ludwig and Summer 1995; Weseloh and Collier 1995; USFWS 1999b). The goals of these private actions are unknown, but they have probably focused on reducing cormorant population levels directly by increasing mortality rates for adults and preventing further growth of populations by disrupting breeding. Shooting adults has also been proposed to prevent colonization at specific sites: NYSDEC was permitted by USFWS to take 30 (1994) to 40 (1995, 1996) adult DCCOs on Bass and Gull Islands on Lake Ontario in conjunction with other colony eradication efforts. No cormorants were reported taken these years, and USFWS subsequently stopped permitting this take because colony eradication had been successful without it (see Lake Ontario, New York (USFWS 1999b), below).

The effectiveness of shooting to kill adults may be limited in some cases. As is discussed in 3) Lethal Control at Aquaculture Facilities (above), DCCOs are wary and difficult to shoot at

aquaculture facilities. During the control program for breeding DCCOs on the St. Lawrence Estuary, Québec (below), Bédard et al. (1999:4) observed,

Depending on the terrain, nest density, and behavior of the birds, the operator could shoot as many as 100 (75 birds/h) from a single position before moving on as the area became devoid of birds. In colonies where forest cover was sparse and / or damaged by prolonged cormorant occupation, the operator could not hide as easily, and lower density of nesters and wariness of the birds reduced success considerably. This situation prevented us from applying a rigorously proportional culling scheme; on some islands, it was impossible to reach the allocated goal while relative tameness of the birds on others led us to exceed it.

The goal of the program was to kill 10,000 adult DCCOs (2,000 / yr * 5 years); culling lasted only four years, however, and in total 7,917 were reported shot. Bédard et al. (1999) also note that adult male DCCOs were more vulnerable to shooting than adult females, with a ratio of 203:100 males to females shot during this control program.

Limited data are available on the effectiveness of killing adults on controlling local populations / colonies. Matteson (1983) notes that shooting DCCOs at pound nets by fishermen did not prevent the nearby Mink Island, Lake Ontario population from increasing from 40 nests in 1945 to 50 nests in 1956. Ludwig and Summer (1995) analyzed leg band recovery data for DCCO colonies in the Les Cheneaux region of Lake Huron; based on the level of immigration into this region, these authors concluded that lethal control of adults would have to be Great Lakes-wide to be effective at controlling local populations (see also Korfanty et al. 1997).

Shooting adults at colonies may have impacts beyond killing individual breeders because of harassment effects on survivors. Ewins and Weseloh (1994) report the shooting of > 50 adults on Pigeon Island, Lake Ontario, in 1993 (when the colony had 818 pairs) reduced reproductive output for that year: fledging rates were 0.3 vs. 1.6 young / nest on a nearby island that was not subject to shooting. Long-term impacts on the colony were not reported.

Killing adults may be a successful technique for controlling regional DCCO populations, but may require that large numbers be killed and geographic range of the program be sufficiently wide. Hobson et al. (1989) noted that illegal killing of adult DCCOs, in conjunction with illegal destruction of eggs and nestlings on Lake Winnipegosis, Manitoba was not enough to prevent this population from increasing. Other investigators have stated that human persecution has contributed to declines in DCCO numbers over large regions during the mid 1900s (Ewins and Weseloh 1994; USFWS 1999b), that decreased shooting as a result of legal protection in 1972 contributed to the subsequent population increases (Carter et al. 1995), or that shooting and other techniques may have slowed the growth of populations in the 1940s and 1950s (Weseloh and Collier 1995). However, it is difficult to separate out the effects of control from the effects of pesticides or changes in fish populations.

The St. Lawrence Estuary, Québec, control program (below) provides an example of a control effort intensive and expansive enough to reverse DCCO population growth. The effectiveness of shooting adults alone may be investigated if the tree-nesting DCCO population behaves independently from the ground-nesting population, as is believed by Bédard et al. (1999). The model used to decide which control techniques were necessary (see PART VI. POPULATION MODELING) predicted 6014 tree nests in 1991 and 4845 in 1993; observed numbers were 3865 and 2555 respectively. Bédard et al. (1999) discuss several possibilities for the higher realized effectiveness of shooting than predicted by the model, and conclude that the unequal sex ratio in adults killed probably accounted for the faster than predicted drop in DCCO numbers.

Effectiveness of killing adults relative to other forms of population control are unknown, although it is believed that, individual for individual, killing adults that have survived to breed is more effective at reducing populations than destroying eggs, nestlings or fledglings. Because first-year birds frequently have low survival rates, a significant portion of control on eggs and nestlings is probably “wasted effort” exerted on individuals that would not have survived to breed anyway. Ludwig and Summer (1995:40) estimate, “From the whole population perspective, killing a young adult just before first nesting will have a 3 to 6 fold greater effect on population control than killing fledglings, chicks or eggs.” Few economic data are available to compare techniques. Bédard et al. (1999) provide a “very rough estimate” for the St. Lawrence Estuary, Québec (below) control program of \$10 / adult shot and \$3 / nest sprayed with oil. Without better economic estimates on the costs of DCCO impacts, the cost effectiveness of population control by any means can not be determined.

Shooting adult DCCOs also raises the potential for by-kill, especially in regions where Great Cormorants exist. For example, on Prince Edward Island a hunting season on DCCOs had no significant impact on this species, but it reduced GRCO populations by 50 %. Nova Scotia does not permit the harvest of DCCOs because of potential impacts to Great Cormorants. Legal shooting of DCCOs in New Brunswick has also led to killing of Great Cormorants there (Milton et al. 1995; Korfanty et al. 1997).

No Control

As in the case of impacts to aquaculture, “no control” should be included on the list of management options for impacts to vegetation and habitat. Not only may the costs of controlling cormorants by all available methods be higher than the benefits of avoided impacts, but also impacts to other species or ecosystems may be better managed by focusing on other impacts than those caused by DCCOs. For example, on the St. Lawrence Estuary DCCOs may be an immediate threat to trees on some islands, and therefore need immediate local controls, but if DCCOs are exacerbating prior over-browsing impacts by herbivores (Bédard et al. 1995), control of the herbivores should also be considered in long-term management plans. As another example, concerns about potential impacts of DCCOs to habitat on the Upper Mississippi River, where cormorants likely altered vegetation prior to European settlement, stem from the fact that the dynamics of the river system and its islands have changed due to regular dredging and development of locks and dams (Kirsch 1995). Managing DCCOs to reduce impacts by other species, including humans, does not solve the original problem, may be less effective than dealing directly with the original problem, and places biologically unjustified blame on the species.

It has also been suggested that landowners be educated about the naturalness of habitat alteration by DCCOs, resolving the problem of DCCO impacts by changing the perception and / or understanding of DCCO impacts (Milton et al. 1995).

Case Studies: Multiple Techniques

The following case studies illustrate the use of multiple management techniques (discussed above) to control DCCO impacts to vegetation, colonial waterbirds and / or fisheries.

Lake Winnipegosis, Manitoba

Interpretations of cormorant control on Lake Winnipegosis in the 1940s and 1950s have varied, even though all citations led ultimately to the description of this program by McLeod and Bondar (1953), who described it as thus:

Control

The cormorant population of the area appeared to be excessive under existing conditions and attempts were made to bring the birds under control without threatening their extinction. To what extent earlier control measures had been applied or had been effective the writers were unable to determine. Certainly the eggs and young had been destroyed sporadically by fishermen for many years. During our visits to the rookeries in 1943, 1944 and 1945 at the height of the breeding season all eggs and young were destroyed.

A recheck of the effectiveness of this procedure on two occasions showed that at least 30% to 50% of the pairs had renested. How successful the attempts at the late season would be is not known but a considerable portion, if not all, the young would be large enough to make the fall migration.

It was decided in 1951 that the killing of the embryos without egg destruction as mentioned by G[r]oss (1944) would prolong the brooding period and remove the stimulus to lay a second time, thus forming a much easier and more effective method of control... (p.9-10)

Control efforts made during visits to two colonies in 1945, and subsequent renesting, are detailed in a discussion of DCCO reproduction. The only other instances of DCCO control or take prior to 1951 presented are observations of egg destruction by lumbermen on Skunk Bay Reef and fishermen on Wade Point Reef, harvest of young from Bachelor Island each year by a mink rancher for mink food, and the collection of “very few” adults for diet analysis.

In 1951 twenty nine different solutions were tested on chicken eggs, five of which consistently killed the embryos. Of these, one solution was applied in the field to DCCO eggs, with an apparent reduction in effectiveness, and the authors concluded that “the method is considered to be not more than 50 % efficient at best” (p.10). No mention is made of further egg treatments or official control after 1951.

Prior to 1943, “information gathered from personal observations and from local fishermen indicated that the Double-crested Cormorant had been increasing quite rapidly” (p.9). Complete surveys of Lake Winnipegosis in 1945, 1950 and 1951 revealed a definite downward trend in the number of observed nests, with 9862, 5949 and 4656 reported, respectively. Based on uncited sources, a correction factor of four was multiplied by the number of nests to estimate the total number of breeding and nonbreeding cormorants on the lake for each year of survey, leading to total population estimates for 1945, 1950 and 1951 of 39,448, 23,796 and 18,624, respectively.

Conclusions relevant to control were (as numbered):

6. If the first eggs or the first young of the season are destroyed at an early age about 50% of the reproducing pairs will renest again almost immediately.
8. Consistent destruction of the eggs and young appears to reduce the breeding population fairly effectively but it was not possible to determine how many were driven to seek new nesting areas or what the mortality from natural causes was.
9. Five different chemical solutions were found to give 100% kill of hen embryos but the only one tried on cormorant eggs in the field gave not more than 50% efficiency.
10. The cormorant population on Lake Winnipegosis is now at a level where the problem in connection with fish predation is less acute than formerly but still requires attention. (p.11)

Conclusions relevant to future population control efforts from the data available in McLeod and Bondar (1954) are limited because it is not possible to determine the level of private control efforts that took place in addition to official ones. The combined efforts appeared to be successful at lowering lake-wide breeding populations, although it is always possible that other unknown factors may have contributed to the decline. Important questions remain unanswered. Would official egg destruction activities have had a significant impact on the population without concurrent private control activities? Because “this decline merged with a much wider fall in numbers retrospectively attributed to pesticides,” (Hatch 1995:13) how much of the observed decline can be attributed to the combined control efforts? If the population had been nearly halved from its peak in 1945, yet still required attention, what population goal would have met the objective of bringing the birds “under control without threatening their extinction”?

Illegal controls have continued on Lake Winnipegosis, and some believe that these disturbances have resulted in *more* breeding on the lake because adults are forced to pioneer new nesting islands which are typically large and treed (B. Koonz, pers. comm.).

St. Lawrence Estuary, Québec

DCCOs nesting on the St. Lawrence River Estuary were subjected to a five-year control effort using both egg oiling and shooting adults to reduce impacts on unique insular forest ecosystems (Bédard et al. 1995a). The program was based on the results of a model of various control techniques (see PART VI. POPULATION MODELING). The goal of the program was to decrease the breeding DCCO population to 10,000 pairs by killing 2,000 adults and oiling 70 % of all accessible nests each year. Although the model was considered conservative, the control program had a much more dramatic impact than was predicted, decreasing the population to approximately 12,000 pairs in only three years.

The control effort was initiated by island landowners concerned about damage to vegetation who lobbied the *Ministère du loisir de la chasse et de la pêche*. Sport and commercial fishermen also voiced complaints about DCCO impacts to fisheries, and while Bédard et al. (1995) suggests that these interests were considered by the *Ministère*, Bédard et al. (1999) specifically state that the fishing lobby was not involved in the decision to control cormorants. The goal of the control program was to reduce the DCCO population to lessen impacts to vegetation, but without eliminating the population altogether: “Maintaining a level of 10,000 pairs in the estuary was deemed adequate to supply opportunities for bird watchers while at the same time maintaining substantial colonies.” (Bédard et al. 1995:79) How this numerical goal was obtained is not explained.

The model (see PART VI. POPULATION MODELING) predicted that neither egg oiling nor shooting adults alone would be successful at meeting the numerical goal within five years, and that even when combined both techniques would have to be applied at what was considered their maximum feasible levels to reach the goal. Therefore it was decided to attempt to shoot 2000 adult DCCOs and oil eggs in 70 % percent of all accessible (ground) nests. Control efforts began in 1989. Shooting was found to be more effective at reducing the tree-nesting segment of the population than was predicted by the model (see Killing Adults, above) and was discontinued after the fourth year. Despite the noted wariness of cormorants, the annual shooting quota of 2000 was apparently not difficult to meet: from 1989 to 1992 the reported number shot was 1993, 2078, 1778 and 2058, respectively. Because access was denied to some oiling efforts, the goal of oiling 70 % of accessible eggs was not met, and ranged from 53 % to 62 % during the five years of oiling efforts (Bédard et al. 1995).

Reasons for the apparent over-effectiveness of the selected control techniques relative to model predictions are discussed in PART VI. POPULATION MODELING. Because impacts by DCCOs were not quantified, the effectiveness of the control program at reducing damages can not be determined in a traditional benefit:cost sense. Whether damage to island vegetation has been reduced, or even avoided, has not been discussed. Bédard et al. (1999) estimate total cost of the control program was \$38,000 / yr, and that, as a very rough estimate, this was about \$10 / adult shot and \$3 / nest oiled. They note, however, that these estimates are site-specific and cannot be generalized.

Long-term effectiveness of the program is uncertain, especially because the population has not been censused since 1993. Measures that could be taken to maintain the population at the desired level (e.g., egg oiling in the largest ground-nesting colonies, disturbances to tree-nesting colonies, dislodging tree nests with long poles) have been discussed, but do not appear to have been modeled or tested (Bédard et al. 1999).

Lake Champlain, Vermont (USFWS 1999a)

Prior to 1999 the State of Vermont Department of Fisheries and Wildlife (VDFW) held USFWS depredation permit MB-751275 to control DCCOs and several gull species through nest removal on Lake Champlain. This permit was used to protect vegetation and wildlife (especially Common Terns) on certain islands from alteration through colonization by these species. In 1999 VDFW requested an amendment to this permit to allow DCCO population control activities intended to reduce impacts to vegetation and wildlife by reducing colonization rates. Requested activities included oiling eggs, killing nestlings and administering avian contraceptives on Young Island, the largest DCCO colony on Lake Champlain; on other islands, permission was requested to continue nest destruction activities. After reviewing the request, USFWS granted VDFW permission to oil eggs on Young Island and to continue nest destruction on other islands. Killing nestlings was not permitted because insufficient evidence was presented that this control was necessary for meeting population goals. Administration of avian contraceptives was not permitted because none was registered for use with DCCOs. The effectiveness of egg oiling was determined through two means: 1) modeling by Fowle (1997) (see PART VI. POPULATION MODELING) and 2) the presumption that reducing local population growth would reduce local colonization rates. Under this control program, virtually all DCCO reproduction was prevented on Vermont waters of Lake Champlain, a roughly 64 % decrease in lakewide reproduction.

Documentation of significant impacts by DCCOs to plant or animal species is difficult to find within the USFWS assessment of the VDFW proposal. Although dramatic changes in plant and avian diversity had occurred on Young Island as a result of colonization by DCCOs, none of the three management options assessed (nest removal, egg oiling and nest removal, and all activities proposed by VDFW) was expected to benefit fauna or flora on this island. On other islands, species of conservation concern were known or expected to exist, but no specific threats had been identified. Even on Bixby Island, where the Canadian Milk Vetch (a State-listed endangered species) was known to exist, no data were presented demonstrating that DCCO colonization attempts occurred on parts of the island where this plant grew. Direct interactions between DCCOs and Common Terns, the other State-listed endangered species mentioned in the assessment, were described as uncertain, and difference in nesting habitat was suggested as a possible explanation for limited interactions between these species.

The USDA/WS environmental assessment prepared in conjunction with the VDFW and NYSDEC for their proposal to the USFWS (Garland et al. 1998) provides slightly more information on DCCO impacts. Cormorant roosting is believed to have shifted gull and tern nesting locations on Popasquash Island and to have “affected” the number of terns using the island. La Barr and Rimmer (1996) are cited after a statement that establishment of a cormorant colony on this island might result in the loss of nesting terns there. Successful colonization of Bixby Island is said to likely result in the loss of Canadian Milk Vetch there, but no details are given. L. Garland, VDFW (pers. comm.) has indicated that the vetch occurs on Bixby Island under the drip edge of cottonwood trees in which DCCOs have attempted to nest.

Past removal of newly-established colonies through nest removal and / or harassment was found to be successful at preventing colonization of new islands on Lake Champlain by DCCOs (see also Nest / Tree Removal, above.). Effectiveness of this technique varied among islands, with some requiring more frequent treatments than others. In general, nest removal was found not to have long-term effects, but required ongoing monitoring and management. Nevertheless, this technique could be used by itself to prevent impacts to vegetation and wildlife.

Effectiveness of egg oiling is more difficult to assess. Based on previous research, ability of this technique to prevent reproduction is well established. Ability of egg oiling to prevent local impacts to vegetation or wildlife is less clear, however. If the Lake Champlain DCCO population received no immigration from other populations, then reductions in this population might lead to reduced colonization rates. However, some level of immigration into the Lake Champlain DCCO population must exist (otherwise, the lake would not have been colonized in 1982), and if high enough, immigration may decrease the effectiveness of local population control at reducing colonization rates. As growth continues in DCCO colonies on New York waters of the lake and in the Missiquoi NWR, and as these colonies become saturated with nesting birds, then colonization rates would be expected to increase. Additionally, holding the lake-wide population below its natural carrying capacity may actually increase the immigration rate if surrounding populations continue to grow to higher densities. Even in the more thoroughly studied European GRCO, factors influencing colonization dynamics are speculative (van Eerden and Gregersen 1995). As with nest removal and harassment, egg oiling requires ongoing management efforts to be effective.

Effectiveness of egg oiling at influencing local population levels is difficult to interpret from the assessment. When discussing effects of egg oiling on this population as predicted by the model constructed by Fowle (1997), the assessment predicts that oiling would not prevent the population from increasing through 2004, but that the rate of growth may begin to decrease after 2002. In response to public comment on the draft assessment, the model is described as predicting that oiling would stabilize the population by 2004. In the ultimate *Finding of No Significant Impact*, egg oiling was predicted to lead to declines in the lake-wide population through 2004.

In conclusion, the addition of egg oiling to Lake Champlain DCCO management efforts will probably not threaten this population. However, because previous nest removal and harassment efforts were successful at preventing colonization, egg oiling was not necessary for meeting VDFW goals (preventing colonization of additional islands by DCCOs). Both management strategies will require ongoing management efforts, and no data were presented to suggest that egg oiling was a more cost effective technique. Without immigration data, effectiveness of population management at reducing colonization rates could not be predicted.

Lake Ontario, New York (USFWS 1999b)

Prior to 1999 the New York State Department of Environmental Conservation (NYSDEC) held USFWS depredation permit MB-828930 to prevent colonization of DCCOs on certain islands in the Eastern Basin of Lake Ontario with nest and egg destruction and lethal control. The goal of these control activities was primarily to protect other colonial waterbirds from negative impacts due to DCCO nesting. In 1999 NYSDEC requested an amendment to this permit to allow DCCO population control activities intended to reduce impacts to colonial waterbirds and to improve the Eastern Basin smallmouth bass population. Requested activities included oiling eggs and shooting adults on Little Galloo Island, the largest colony on the Eastern Basin; on other islands, permission was requested to continue nest destruction activities. After reviewing the request, USFWS granted NYSDEC permission to oil eggs on Little Galloo island and to continue nest destruction on other islands. Shooting of adult birds on Little Galloo was not permitted “because insufficient information is available to justify this take in accordance with USFWS depredation policies.” Because USFWS is not permitted to issue depredation permits to benefit sport fisheries, the decision to permit egg oiling was based on impacts to birds and vegetation. Under this control program, all DCCO reproduction was permitted to be prevented on U.S. waters of the Eastern Basin, a roughly 75 % decrease in reproduction.

In addition to legal NYSDEC control efforts, this population was subject to illegal control activities in 1998, during which approximately 2000 DCCOs, mostly juveniles, were shot on Little Galloo Island. Disturbance by researchers was cited as the probable cause of observed declines in DCCO colony size on Little Galloo Island since 1996.

More detailed biological information is provided in this environmental assessment than in the VDFW assessment (above), but specific impacts are still difficult to find. Although dramatic changes had occurred on Little Galloo Island as a result of colonization by DCCOs, none of the three management options assessed (nest destruction, egg oiling and nest destruction, and all activities proposed by NYSDEC) was expected to benefit fauna or flora on this island. On other islands, species of conservation concern were known to exist, but no specific threats were identified. Common Terns were State-listed threatened; however, impacts of DCCOs on ground-nesting terns in general were described as minimal, direct interactions between DCCOs and Common Terns were uncertain, and difference in nest-site selection was suggested as possible explanation for limited interaction between these species. Black-crowned Night-herons were described as rare and local nesters in northern New York, and past interactions with DCCOs suggested cormorants could exclude BCNHs from some sites, but DCCOs impacts on regional BCNH populations were unknown. Caspian Terns were also mentioned within the assessment, but no negative interactions had been documented between this species and DCCOs (even on Little Galloo Island where both species were increasing), and negative impacts from DCCOs were predicted on Little Galloo Island only if the Caspian Tern colony grew to an even larger size.

Past removal of newly-established colonies through nest removal was found to be successful at preventing colonization of new islands on the Eastern Basin by DCCOs. As on Lake Champlain (above), effectiveness of this technique varied somewhat among islands, and the effects were not long-term, but in an ongoing monitoring and management program this technique could be used by itself to prevent impacts to vegetation and wildlife.

Assessment of the effectiveness of egg oiling at reducing population levels and colonization rates is similar to that of egg oiling on Lake Champlain (see above). Immigration data are not available for this population. Predictions of effectiveness of egg oiling at managing

populations in this assessment appear to be based on the Lake Champlain model by Fowle (1997), and in this assessment, these predictions showed the same pattern of multiple interpretations as in the Vermont assessment.

In conclusion, addition of egg oiling to Eastern Basin DCCO management efforts will probably not threaten this population. However, because previous nest removal efforts were successful at preventing colonization, egg oiling was not necessary for meeting NYSDEC goals of preventing impacts to vegetation and wildlife. Both management strategies require ongoing management efforts, and no data were presented to suggest that egg oiling was a more cost effective technique. Without immigration data, the effectiveness of population management at reducing colonization rates could not be predicted.

Conclusions: Managing Impacts to Vegetation and Habitat

In cases where DCCO nesting activities may impact threatened or endangered species in habitat-limited regions, prevention of colonization has been demonstrated as a successful tool for preventing these impacts without resorting to population control. If cases are found where these local control techniques are insufficient, population control of nesting DCCOs may be successful at reducing population levels under two conditions: (1) scale of control is large enough to overcome the effects of immigration and (2) the control effort is well coordinated, long-term, and sufficiently rigorous to overcome density-dependent compensation mechanisms. Few data are available to predict what levels of control are needed to be successful, however. Modeling of European GRCO population control techniques suggests that preventing the establishment of new colonies is the only technique not buffered by density-dependent mechanisms (Bregnballe et al. 1997); currently this population control technique is being utilized in Denmark (Bregnballe and Asbirk 1995).

MANAGING IMPACTS TO FISHERIES

Site-specific DCCO impacts to fisheries generally occur in three situations in which fish are aggregated at high densities: 1) large-scale releases of hatchery-raised lake trout and other lake fish, 2) salmon smolt runs, and 3) fish harvest methods such as pound nets and weirs that congregate fish in enclosed areas that are open to cormorant predation. Management techniques for these impacts are discussed below.

Techniques for managing entire DCCO populations are available and may be successful at reducing DCCO numbers if applied on the proper scale (see Managing Impacts to Vegetation and Habitat, above). However, because many uncertainties remain regarding impacts of DCCOs on fisheries (see PART III: DIET), effectiveness of population reductions at reducing impacts to fisheries can not be predicted.

Hatchery-raised Trout Releases

Various simple techniques are available for preventing released trout from remaining in large aggregations that are vulnerable to preferential predation by cormorants. Fish can be released during the night, providing them with time to disperse before cormorants begin feeding the following morning. Fish dispersal can also be aided by releasing them from barges in deep water, rather than from the shore. Studies in 1993 and 1997 by the NYSDEC found that barge-stocked trout survived better than those stocked from shore. Fish releases in streams may be

improved by stocking earlier in the season, before cormorants return from migration (NYSDEC 1999, NYSDEC Special Report 1999).

The criticism by Conniff (1991) that night releases were unnecessary on Lake Ontario because DCCO diet was found to be primarily alewives in that lake appears to overlook the temporal differences in diet studies versus stocking events. Although the proportion of trout to alewives in DCCO diet may be small over an entire season, the proportion could be much higher on days that trout are stocked if cormorants feed preferentially on these fish. If such a change in feeding behavior were observed in DCCOs, and if this feeding behavior has a significant impact on the survival of the released trout, changing to night releases would be beneficial.

Lethal and / or non-lethal harassment may also reduce predation on swarms of recently released fish. As is discussed in the Lake Ontario, New York (USFWS 1999b) case study, NYSDEC has held permits to shoot up to 40 DCCOs at fish stocking locations 1997-1999. However, no DCCOs have been reported taken under this permit, suggesting that the above alterations to stocking methods may be successful without harassment (USFWS 1999b).

Salmon Smolt Releases / Runs

Bayer (1989) presents a detailed review of salmon hatchery management techniques that may help reduce losses of smolts to predators in general on the Pacific Coast. Most investigators have addressed the topic of DCCO predation on salmon smolts, focusing on techniques for relieving DCCO predation pressures only. These DCCO-specific techniques all attempt to move migrating smolts past areas of cormorant predation (especially river segments near DCCO colonies) as quickly as possible and with smolts highly dispersed. Milton et al. (1995) and Price and Nickum (1995) both cite personal communication with G. Farmer that changing the time of releases improves survival of smolts. Releasing during times of higher river discharge rates and later in the year appears to reduce recovery rates of salmon tags at nearby DCCO colonies. Releasing smolts higher upstream from regions near DCCO colonies or above lakes upstream from cormorant feeding areas also breaks up congregations of smolt that might otherwise attract cormorant predation. Matteson (1983) citing Bissonette et al. (1982) and Roby et al. (1998) also suggests altering the timing and location of salmon releases.

Nets / Weirs

Although largely anecdotal, data exist that strongly suggest cormorants may have an impact on fish catches in purse-seine nets, pound nets and weirs. These are all fish-capture techniques that passively trap fish in enclosures that are open to the sky, rendering them vulnerable to avian predation until the fish are harvested. Although direct losses through predation may be of concern, the impact discussed in the greatest detail appears to be gilling: behavioral responses to cormorant feeding by fish (even those too large to be eaten by DCCOs) may result in the fish gilling themselves in nets. Gilled fish have a lower market value than non-gilled fish (Wilson 1941; Omand 1947; Matteson 1983; Craven and Lev 1987). Craven and Lev (1987) reported that an increase in the DCCO population on the Apostle Islands, Lake Superior was associated with increased gilling rates, and that increased gilling rates were also associated with fledging DCCOs first beginning to forage for themselves. Site-specific, non-lethal control techniques appear to have been discussed most frequently in the literature. Population control was considered in the Apostle Islands, but not attempted because the colonies involved were on National Park Service lands, where such control was not permitted.

Equipment Modification

Covering these nets with netting or wires has been proposed to prevent DCCO access to fish trapped within. However, DCCOs appear to adapt readily to entering the nets through the same entrances as the fish, easily exiting by the same route and leaving gilled fish behind (Omand 1947; Matteson 1983). Modifying nets and the poles that support them to prevent perching has been suggested not only because these perches might serve as vantage points for foraging cormorants, but also because the mere sight of predators may frighten fish into gilling themselves (Matteson 1983; Craven and Lev 1987). For pound nets, metal cones or spikes have successfully prevented perching on poles, and electric wire has prevented perching on the nets themselves (Craven and Lev 1987; S. Craven pers. comm.). Reducing the size of the mesh in nets has been suggested as a means to reduce gilling (Matteson 1983), but this technique appears not to have been investigated.

Harassment

The harassment techniques suggested and tested at nets are similar to those used at aquaculture facilities, as are the conclusions drawn from them. Craven and Lev (1987) tested the effectiveness of owl decoys, mylar balloons, hanging scarecrows, untended boats, and an audio device called Av-Alarm. They found that DCCOs habituated to all devices over time, but that the combination of a boat with a scarecrow was successful at preventing cormorants feeding in the nets for at least five weeks of a six week trial. Success of the techniques was generally measured in terms of numbers of cormorants observed feeding in the nets, but the combined scarecrow and boat technique was reported to have reduced gilling rates as well. The Av-Alarm not only was unsuccessful in that DCCOs were observed perching within seven feet of the speaker, but also was incompatible with National Park Service policies. Craven and Lev (1987) ultimately suggest a combination of harassment (boat, scarecrow, mylar balloons) and equipment modification (metal cones or spikes on poles and electric wires on nets).

Matteson (1983) also discusses harassment techniques that were tried by fishermen themselves in the Apostle Islands, none of which were reported to have worked: rubber snakes, wind wheels, brightly-colored flags, eagle decoys, pieces of metal, and gas cannons. Not only was the gas cannon incompatible with National Park Service policies, but it also was reported to keep fish away from the nets. Matteson (1983) also mentions techniques that were suggested, but apparently not tested: gas cannons, sonic horns, distress calls, hanging dead cormorants from posts, and suspending Bald Eagle or Gyrfalcon decoys over nets in conjunction with recorded vocalizations of these birds.

Wilson (1941:11) reports discussions with Door County, Wisconsin fishermen who “claimed that after the cormorants appeared in late summer no fish were caught in the pound nets. Where catches of several hundred pounds of fish had been made, lifts dropped to 15 – 20 pounds. In desperation scare crows were nailed onto the net stakes to frighten the cormorants. Good lifts resulted.”

In Nova Scotia cormorant predation was described as a site-specific problem for weir and gill-net harvests of Mackerel, Atlantic herring and alewife. Scare and kill permits are reported to have resolved the problem of cormorant predation, though details are not presented (Milton et al. 1995; Keith 1995).

Alternative Perches and Prey

Craven and Lev (1987) note that DCCOs perch on poles used to support pound nets even when nets are not in place (natural perches being scarce in the area), and suggest that erecting perches away from nets, but still in suitable foraging habitat, may help control efforts by “diluting the problem” and by providing cormorants with alternative perches from those at which harassment is being practiced. These alternative perching sites could be combined with the suggestion by Matteson (1983) that DCCOs be provided with an alternative foraging site stocked with potential prey near the main Apostle Islands colony. Matteson (1983) admits, however, that supplemental feeding may be a short-term solution that ultimately leads to larger long-term problems associated with population growth.

Reduce Time Fish Spend in Nets / Weirs

Reducing the amount of time that fish spend in nets and weirs would reduce their exposure to DCCO predation and presumably impacts to these fisheries (Matteson 1983). No economic analyses have been reported, however, that compare the costs of increased harvesting frequency relative to avoided losses due to cormorants.

4) Conclusions: Managing Impacts To Fisheries

Effective techniques are available for reducing local DCCO impacts to fisheries through changes in management techniques, equipment modification and harassment. Reducing DCCO populations requires large efforts to be effective (see Managing Impacts to Vegetation and Habitat, above) and may not result in significant benefits to fisheries populations if predation is compensatory (Bayer 1989; Marquiss and Carss 1997).

SURVEY RESULTS

Only 11 of the 37 states and provinces for which surveys were completed reported that cormorant control methods were currently being employed (Table 28). Of these, all employed lethal and non-lethal measures except Québec, which employed only lethal control, and Massachusetts, which only employed non-lethal control. Previous control efforts (shooting and egg destruction) were reported for Rhode Island, but the program was discontinued due to a lack of personnel to conduct work.

Six of the states employing control measures were in the southern U.S. These states were conducting control programs because of depredations at aquaculture facilities and fish hatcheries. All of these states incorporated lethal and non-lethal control measures.

In the Northeast, New York and Vermont are employing control measures due to habitat destruction and impacts to other colonial waterbirds in the Great Lakes. Massachusetts has undertaken limited control measures at specific sites in the 1990s. At Ram Island, a tern restoration project was underway and non-lethal cormorant control was employed in 1990 and 1991. Efforts were described as a “worrying program”: 2 individuals went to the island everyday and harassed nesting DCCOs, broke up their nests and destroyed eggs. These activities were very successful in discouraging DCCOs from nesting. In 1991, activities were begun on April 29, and all birds abruptly abandoned the site on June 7; they have not since returned. Other measures had previously been employed at this site, including the installing of a bathel—a 4 foot wire fence run throughout the colony area, and topped with strands of twine. This was done to make takeoffs and landings more difficult, but was a total failure. Birds perched on the stakes,

incorporated twine into their nests, and braced their nests against the fencing. Harassment is being employed at a night roost/staging area in Orleans, MA and includes pyrotechnics. Virginia did not provide information about why cormorant control was employed.

In Canada, management of cormorants falls under provincial jurisdiction. The government of Manitoba does not undertake any control activities, but much illegal control occurs around Lake Winnipegosis. These activities are undertaken principally because cormorants are perceived to impact fish populations, even though a recent study by Hobson et al. (1989) indicated that commercially valuable species do not contribute significantly to the diet of cormorants breeding on Lake Winnipegosis. In fact, data collected between 1945 – 2000 indicate declines in commercially valuable fish populations on Lake Winnipegosis are due to overfishing (W. Lysack, pers. comm.). Cormorants are also killed in this area because of impacts to trees, but to a lesser degree. Québec undertakes cormorant control because of impacts to unique insular habitats and privately owned islands (see above).

Table 28. States and provinces practicing Double-crested Cormorant control.

State	Lethal measures	Non-lethal measures
AL	Shooting	Harassment
AR	Shooting	Harassment, noise-making, decoys
LA	Shooting	Multiple harassment techniques
MA	None	Harassment
MS	Shooting	Harassment; Night roost dispersal program
NY	Egg destruction, egg oiling	Nest destruction
OK	Shooting	Hazing
QC	Shooting, egg oiling	none
TX	Shooting	Harassment
VA	Yes ¹	Yes ¹
VT	Egg oiling	Harassment; nest destruction

1 Both lethal and non-lethal measures are undertaken, but details on specific measures employed were not provided.

PART VI. POPULATION MODELING

INTRODUCTION

In his overview of the symposium “The Double-crested Cormorant: Biology, Conservation and Management” held in 1992, Nisbet (1995:248) provides an excellent summary of knowledge of DCCO population dynamics and modeling efforts:

In comparison with other widespread and common colonial waterbird species, the breeding biology and population dynamics of the Double-crested Cormorant are very poorly known. The comparative neglect of this species by biologists may be due in part to prejudice against an ugly, black, smelly bird (Hatch 1995), but is more attributable to difficulties in studying it. Cormorants are very sensitive to disturbance when nesting, commonly nesting in trees, and are difficult to catch or to observe in winter. With one notable exception in California (M. M. Bishop, unpubl. data), no long-term studies of breeding birds were presented or cited [in the symposium], and no studies of individually-marked or known-aged birds have been conducted. Although data on productivity have recently been generated (Weseloh et al. 1995, Chapdelaine and Bédard 1995, Stenzel et al. 1995), there is little or no information on age at first breeding, recruitment, dispersal or age-specific mortality. Consequently, the parameters used in population models (Bédard et al. 1995) are quite speculative. More important, we do not know how these population parameters vary with population density, colony size or environmental conditions. Hence, we do not know what factors may limit the populations, either now or in the future. Data presented in several papers suggest that some populations may have started to level off in the late 1980s. However, even if this proves to be true, it is not clear whether the limiting factors would be those acting in the breeding area or in the winter quarters. At present, there is no way to predict population trends even in the immediate future (five years or less). The appearance and rapid spread of Newcastle disease among breeding cormorants in 1992 increases the uncertainty about future trends.

The Birds of North America account for DCCOs (Hatch and Weseloh 1999:20) indicates that there has been little improvement in knowledge of DCCO life table parameters since the above-mentioned symposium, stating,

Compared to other widespread colonial waterbirds, population dynamics of Double-crested Cormorant poorly studied; no life table constructed. Key demographic parameters have been established at only 1 colony, in a 3-yr study of *P. a. albociliatus* on Mandarte I., British Columbia, where numbers were increasing by 8.4% / yr (van der Veen 1973). Large demographic differences are likely between populations that are resident or migratory, expanding or stable, but such differences have not been examined.

Their Appendix 2 cites 16 sources of DCCO annual reproductive success estimates, ranging from 0.01 young / nest (Gress et al. 1973), to 2.59 young / nest (McNeil and Léger 1987); the most recent of these estimates came from Bédard et al. (1995). When discussing annual reproductive success, Hatch and Weseloh (1999:20) also caution, “All studies probably are subject to observer effect and subsequent predation...and all figures are much lower for DDE-contaminated populations.” Discussion of population regulating / density dependent factors remains speculative (e.g. Ludwig and Summer 1995). Despite lack of information on DCCO life table parameters, informative models can be built to investigate cormorant population dynamics and to determine which parameters need to be obtained with greater accuracy to reduce uncertainty in predictions of population trends.

Many types of population growth models exist. Some are deterministic (i.e., all parameters are fixed values and therefore models predict the same outcome every time). Other models are stochastic (i.e., some or all parameters vary, causing the model to behave differently with each trial). Deterministic models are sometimes subjected to sensitivity analyses (in which parameters are varied and observations are made of subsequent changes in model behavior) because life history parameters are either poorly known or have been shown to vary. These analyses are important for determining how much confidence can be placed in predictions made by deterministic models. Stochastic models, in contrast, incorporate variation internally, and statistical analyses are performed on the models' multiple trials to measure overall levels of variation. To our knowledge, all DCCO models have been deterministic. Models can also be classified as exponential (parameters are constant/density independent) or logistic (parameters are variable / density dependent:). Not all density dependent models behave logistically, however. Other population model types exist, but will not be discussed here. Also, we do not review non-population growth models, such as DCCO bioenergetics models (e.g. Cairns et al. 1991, Glahn and Brugger 1995, Madenjian and Gabrey 1995).

Selection of the proper model to construct depends on data available, questions the model is intended to answer, and external factors likely to influence population dynamics over time period to be modeled. For example, although exponential models are biologically unreasonable in their assumption that populations are not limited by food, physical space, disease, or other factors, for many applications they 1) may be the only models possible to construct given available data, and 2) may approximate the dynamics of populations well below their carrying capacity. Additionally, the scale of the population to be modeled is an important consideration, especially for predicting the effects of emigration / immigration.

For DCCOs, models have generally been constructed to perform one of three tasks: 1) estimate past growth rates, 2) predict future DCCO population levels (and presumably impacts) in the absence of control, and 3) predict the effects of various management techniques on DCCO population levels. Another interesting use of a population model (see Lake Ontario, below) attempts to determine whether observed growth in the Lake Ontario DCCO population could have occurred without immigration.

In this section we review the basic population models available, focusing on those used for DCCOs, and discuss case studies where appropriate.

DETERMINISTIC EXPONENTIAL MODELS

Basic Exponential Growth Models

The basic exponential growth model is one of the simplest models of population dynamics. Essentially, this model assumes a constant per capita rate of increase (or decrease) over time, which results in populations growing (or declining) by a fixed percentage of their size. Modeling specific populations involves measuring past growth rates and then applying these rates to current population levels to project future growth. There are two basic equations, one used primarily for discrete time intervals (usually years):

$$N_t = N_0 \lambda^t$$

and one for continuous time:

$$N_t = N_0 e^{rt}$$

where N_0 is initial population size, N_t is predicted population size at time t , λ is the annual population growth factor (or annual finite rate of increase) and r is the instantaneous, per-capita growth rate (or simply rate of increase) of the population. The relationship between these growth measurements is $e^r = \lambda$, and $(\lambda - 1) * 100$ is the mean annual percent increase in the population. The instantaneous growth rate, r , incorporates the four factors that can change population size: birth, death, immigration and emigration. Thus, given an initial population size and that population's past growth rate, it is possible to estimate future population levels if that growth rate remains constant by simply plugging the numbers into these equations. Predicting how long DCCO population growth rates will remain constant (unaffected by density-dependent factors such as food, space, disease, etc.), and therefore how far into the future populations growth can be reliably predicted, is difficult because of the lack of data for DCCOs.

Of the many studies that report exponential growth rates, relatively few have used this model explicitly to predict future DCCO population trends. Those that have done so have limited their projections to relatively short lengths of time. USFWS (1999a,b) project DCCO population growth on Lake Champlain, Vermont and on the Eastern Basin of Lake Ontario, respectively, over one year to assess the impact of proposed control measures on these populations. Fowle (1997) predicts growth in the Lake Champlain population over four years (1996 – 2000) in the absence of control. Bédard et al. (1995) predict population growth in the St. Lawrence Estuary population over five years (1987 – 1992) in the absence of control.

If a given population's size is known for two points in time (N_0 and N_t), the above equations can be used to estimate the average growth rate of the population, under the assumption that the population grew exponentially during this interval. Solving for either r or λ produces the following equations:

$$r = \frac{\ln N_t - \ln N_0}{t} \quad \text{and} \quad \lambda = (N_t / N_0)^{1/t}$$

These numbers can easily be converted into the often-cited “annual percent increase” format, as described above. We note that some investigators appear to have reported r as annual percent increase (e.g., Ewins and Weseloh 1994; Bédard et al. 1995; Weseloh et al. 1995). While the two numbers are similar for most common growth rates, growth rates for DCCO populations have often been measured at 20 % or more, and in these cases the difference is more significant (Figure 89).

Because growth rate estimates calculated from the simple exponential model are based on only two data points, they should be approached with caution. Anyone using the exponential model to calculate a population's growth rate makes the assumption that that population grew exponentially between the two selected points, even though the population may have made the transition linearly, sigmoidally or by any other trajectory. Some argument ought to be made to justify the selection of the exponential model over others when describing population growth. Otherwise, the impression may inadvertently be given that the population is growing exponentially, when in fact there are no data to indicate this. The Pigeon Island, Ontario DCCO colony (Ewins and Weseloh 1994) illustrates this point: while population growth rates were reported in terms of annual percent increase (based on the exponential growth model and nest counts for 1978 and 1993), nest count data for the years in between reveal that population growth

was more linear than exponential, adding approximately 57 nests each year on average (Figure 90). This case study also illustrates another potential problem with the use of the basic exponential model for calculating growth rates: due to variation in population growth rates between years, the overall rate calculated may vary considerably depending on the two years selected for analysis. In this case study, annual percent increase calculated from nest counts in 1978 and 1991 is 52 % ($r = 0.420$), whereas calculated from 1979 and 1990 data it is 24 % ($r = 0.216$).

Figure 89. The relationship between annual percent increase and r .

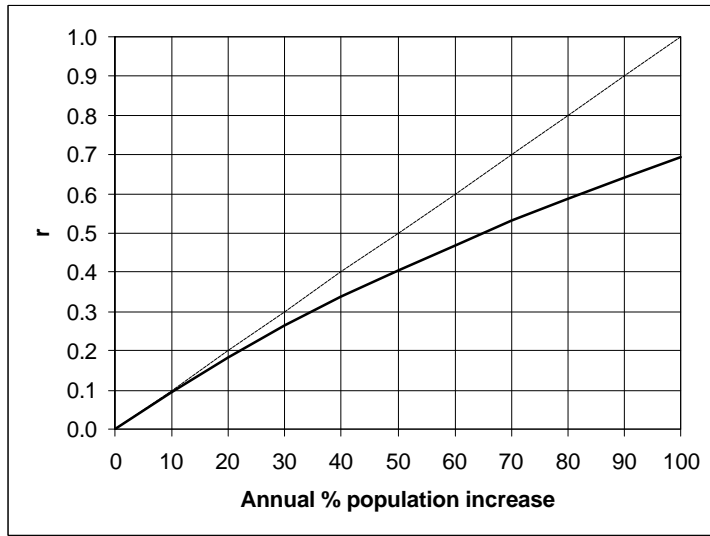
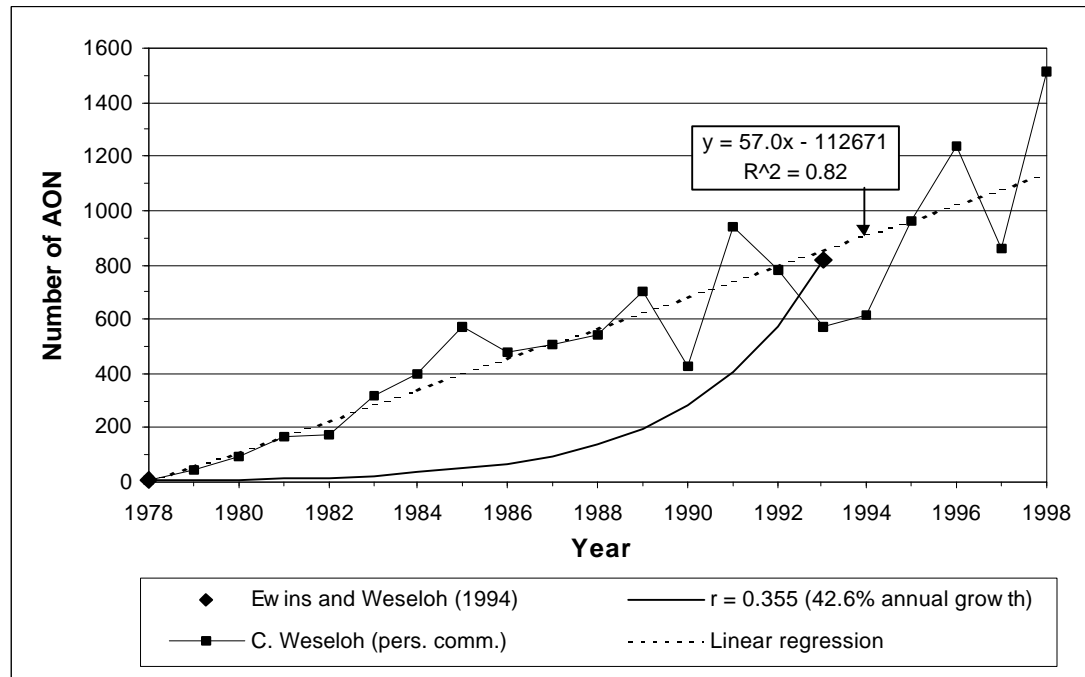


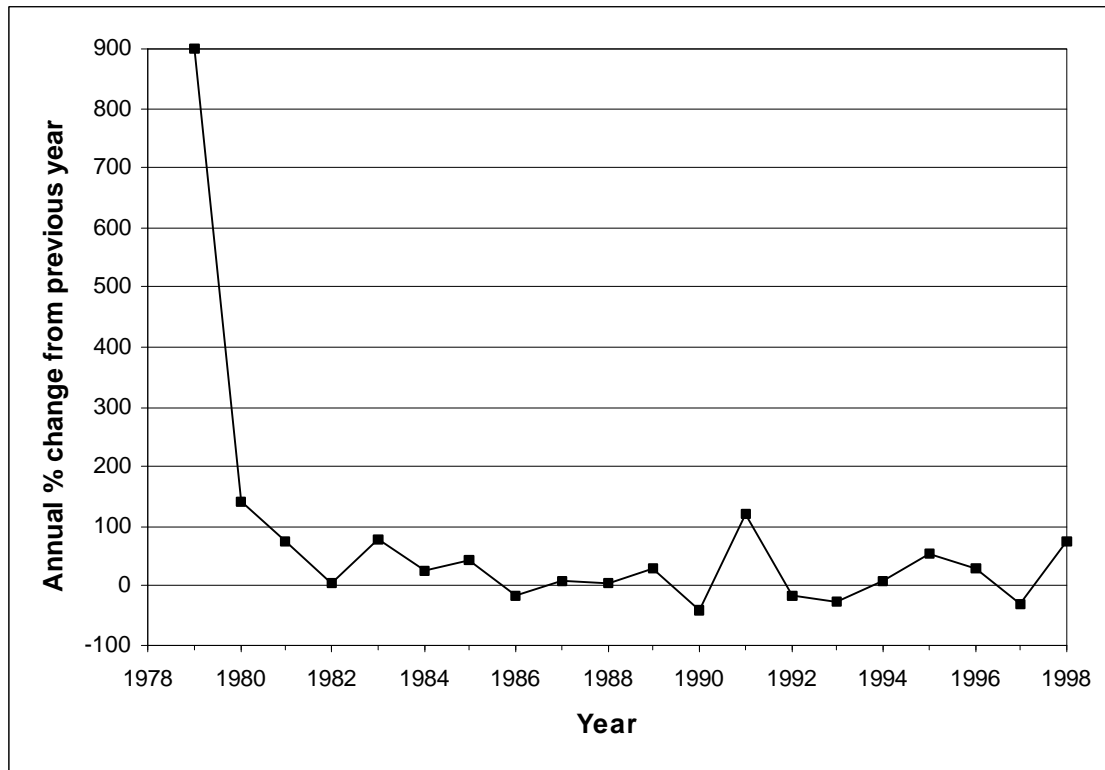
Figure 90. Numbers of Apparently Occupied Nests (AON) on Pigeon Island, Ontario with population growth estimates.



Because of these potential problems, this form of the exponential growth model should not be used to calculate growth rates when data are available for more than two years. To do so ignores valuable data that may reveal population trends other than exponential growth and also wastes the often considerable efforts made to gather these data. Instead, techniques that incorporate all available data, such as regression analysis, should be used (e.g. Matteson et al. 1999).

One additional complicating factor is the contribution of immigration to population growth and the effect it may have on projections of population growth into the future. Initial growth of a newly established colony or population may rely more on immigration than on reproduction. For example, the Pigeon Island, Ontario colony grew from four nesting pairs in 1978 to 40 pairs the next year, an increase of 900 % (Figure 91). For DCCOs, such growth is obviously not possible by reproduction alone, even if all fledglings survived to reproductive age, and therefore must have included a large immigration component. This large second-year immigration event had a considerable impact on the overall growth rate calculated in this study: whereas annual growth from 1978 (four nests) to 1993 (818 nests) was 43 % based on the exponential model, making the calculation with 1979 (40 nests) and 1993 data reduces the annual growth rate to 24 %. Because the first growth estimate was influenced by a large initial immigration rate (relative to colony size) that has subsequently decreased, this estimate would not be appropriate for predicting future population growth.

Figure 91. Annual percent change in number of apparently occupied nests (AON) in Pigeon Island, Lake Ontario.



Age-Structured Exponential Growth Models

Age-structured models attempt to replicate natural population dynamics more accurately than basic exponential models by incorporating within them differences in parameters that exist between age (and sometimes sex) classes. These models predict exponential growth (or decline) because they assume that all parameters are constant (density independent) for their respective age and sex classes. Therefore, these models are similar to the basic exponential model in the number of years they can reliably predict future population trends.

Differences in mortality rates between young of year, subadults and adults appear to be most commonly included in DCCO models (see case studies, below). As is discussed in the following case studies, interpretation of age-structured models, when age-specific parameters are poorly known, is difficult and investigation of the confidence levels of model predictions is important. Differences in mortality rates between the sexes may be modeled, but this does not appear to have been included in any DCCO models. Age-specific fecundity rates are not known for DCCOs and appear not to have been modeled. Other poorly known parameters such as immigration and emigration have not been included in age-structured DCCO models.

We found only three age-structured DCCO population models. Bédard et al. (1995; St. Lawrence Estuary, Québec, below) and Fowle (1997; Lake Champlain, Vermont, below) constructed models of this type to investigate the effectiveness of various DCCO control options. Even though few age-specific data are available for DCCOs, age-specific models were necessary in both of these studies to assess effectiveness of egg oiling and / or shooting adults at manipulating population levels. Each study used different techniques to estimate age-specific parameters, to establish the initial age class distribution (unknown for both studies), and to analyze model output. Only Fowle (1997) performed sensitivity analyses of model output. The third age-structured model (Price and Weseloh 1986; Lake Ontario, below) did not investigate DCCO control, but rather the question of whether observed growth in the Lake Ontario population (1974 – 1982) could have occurred exclusively through reproduction of Lake Ontario colonies or whether immigration was necessary to produce observed population growth rates. This study used yet another technique for estimating population age structure. All three studies assumed all breeding-age adults nest every year, an assumption that has not been investigated in the field for DCCOs. In contrast, van Eerden and van Rijn (1997) estimated that 40 % of potentially available breeders (> 3 yrs) in the Oostvaardersplassen, The Netherlands, GRCO population did not breed (see also PART II. POPULATION DATA AND TRENDS, Non-breeding Birds and Total Population Size).

Lake Ontario

Price and Weseloh (1986) constructed an age-structured, deterministic, exponential model to investigate the question of whether observed 65 % annual growth in nesting DCCOs on Lake Ontario (1974 – 1982) could have occurred without immigration. The model included age-specific mortality rates, equal fledging rates for all breeding birds (i.e., no age-specific fecundity), age of first breeding at either two or three years, and all adults breeding each year. Not discussed are model structure in terms of year or life-stage classes. Initial age class distribution appears to have been set at 22 adult males and 22 adult females (based on 22 nests observed in 1974) with pre-breeding age classes based on assumed fledging and survival rates for nestlings from 22 nests each of the previous few years (and equal sex ratios).

Initially, the model was run under conditions that led to a stable population: first breeding at three years, annual fledging rate of 1.0 young / nest, 70 % pre-breeding mortality and 15 %

annual adult mortality. Pre-breeding mortality rates appear not to have been calculated annually, but for the entire period between fledging and first breeding at either years two or three. Under these conditions annual growth rate was 0 %.

Given hypothesized changes in model parameters that might occur during periods of rapid population increase (as this population appeared to be exhibiting) the model was re-run with altered parameters: breeding at two years, annual fledging rate of 2.5 young / nest (based on unpublished data averaged over the Great Lakes), pre-breeding mortality of 50 % and 10 % annual adult mortality. This model produced an annual growth rate of 37 %, considerably lower than the observed 56 % annual increase. Parameters were adjusted again until the goal of reproducing 56 % annual growth was met. One successful combination consisted of breeding at two years, 2.8 fledglings / nest / year, 31 % pre-breeding mortality and 10 % annual adult mortality. The other successful combination consisted of breeding at two years, 3.0 fledglings / nest / year, 40 % pre-breeding mortality and 5 % annual adult mortality. These parameters were considered plausible for DCCOs, suggesting that immigration was not necessary for the observed 56 % annual growth between 1974 and 1982.

To further investigate the data, annual percent increases were calculated for years between 1974 and 1982, and these levels of increase were compared to estimates of numbers of young that might have entered the population from breeding two years prior (assuming breeding at two years, 2.8 young / nest, 31 % pre-breeding mortality, 10 % annual adult mortality, all breeding-age adults breed). According to this analysis, observed growth in years 1976, 1978 and 1979 (171 %, 110 % and 56 %, respectively) was higher than predicted based on estimated reproduction in previous years, and it was concluded that in these years “substantial migration (up to 55 % of the year’s growth) must have occurred, unless other colonies had existed to produce the extra young two years earlier.” However, in discussing this model, Duffy (1995) notes that the population may contain non-breeding adults that recruit into colonies during “exceptional years” and that not necessarily all of the increase in the population beyond that predicted by recruitment needed to have been due to immigration alone.

St. Lawrence Estuary, Québec

Impacts by DCCOs to vegetation in the St. Lawrence estuary were considered to be high enough that the population should be reduced. The goal was to reduce the population to 10,000 nesting pairs from a high of 14,662 in 1987 (see also St. Lawrence Estuary, Québec in PART V. MANAGEMENT OPTIONS). Bédard et al. (1995) constructed an age-structured, deterministic, exponential growth model to determine what level of control would be needed to meet this goal. The two techniques selected for control were oiling eggs to kill embryos (a technique only feasible for ground-nesting colonies) and shooting adults (a technique generally limited to tree-nesting colonies). Although investigators wanted to avoid killing adults because public opinion does not favor this technique, the model predicted that only a combination of the two techniques would be successful at meeting the population goal and both were used.

The model was constructed with POP-II software (Fossil Creek Software 1990). Number of age classes was not reported. Four pieces of information were needed to construct the basic model: 1) age at first breeding, 2) age-specific mortality rates, 3) fledging rate and 4) population age structure. Age at first breeding was assumed to be three, although it was acknowledged that some two-year old DCCOs may breed. Though not explicitly stated, it appears that all birds ≥ 3 yr were assumed to breed. Initial age-specific mortality rates were taken from van der Veen (1973). The initial fledging rate of 2.0 fledglings / female / year appears to have been arbitrarily

chosen from a range of reported values and was applied equally to all breeding-age birds. To estimate population age distribution, a model was constructed with the above three parameters and a simulated founding population of 100 males and 100 females. The model was run for 24 years and the age class distribution at that point was used to represent the age-class distribution in the St. Lawrence population. Why this technique was used, and why 24 years was selected was not discussed. The above combination of parameters could not reproduce a stable, let alone growing, population, so the first three parameters were altered until a combination was found that successfully reproduced the estuary's observed growth of 6155 pairs in 1979 to 14,662 pairs in 1987. The final combination of parameters consisted of first breeding at three years, 50% year 1 mortality, 25 % year 2 mortality, 20 % mortality in years 3 and above, and a fledging rate of 2.0 juveniles / female / year. The resulting parameters were then changed to simulate the various control techniques. Throughout modeling, tree-nesting and ground-nesting components of the population were assumed to be independent of each other and no birds moved between them from year to year. The above parameters were applied equally to both populations. Immigration into and emigration from the St. Lawrence Estuary were assumed not to occur.

The maximum level of egg oiling considered feasible was 70 % of all ground nests. This control method was presumably simulated by reducing breeding output in 70 % of the ground-nesting birds to zero, but details are not presented. The oiling of 50 % of all ground nests was also modeled. An unstated assumption of this model is that none of the adults in nests subjected to oiling re-nested and that oiling was 100 % effective at killing embryos. Although evidence is presented that the latter assumption is valid, little is known about the activities of the roughly 50 % of DCCOs that abandon oiled eggs (see Egg Oiling / Spraying in PART V. MANAGEMENT OPTIONS).

Modeling effects of shooting adults was more complex. It was assumed that 1) shooting one individual from a nesting pair would cause their nest to fail, 2) 75 % of adults shot would belong to different pairs (thus, 25 % of the shooting was "wasted effort" with respect to reducing reproduction), and 3) nest predation by gulls, crows and ravens as a result of disturbance during shooting episodes would cause an additional 15 % reduction in reproduction. No empirical data were available for these three assumptions. The combined effect of these impacts on recruitment was calculated to be an overall reduction of 40 % at colonies where shooting was to take place (15 % due to direct losses, 15 % through predation losses, and 10 % through loss of a breeding partner). This effect was modeled by reducing fledging rates to 1.2 young / female / year under a 2000 / year culling strategy and 0.92 young / female / year under a 3000 / year culling strategy. These calculations implicitly assumed culling would have an equal effect on both sexes.

Results of modeling control strategies in different combinations showed the overall DCCO population continued to increase when either egg oiling or shooting adults was practiced separately. Only combination of both techniques was reported to be successful. Results are difficult to interpret, however, as some figures and captions in Bédard et al. (1995:81-83) do not match. Furthermore, no sensitivity analyses were performed to determine how uncertainty in model parameters might have influenced these conclusions. From these results investigators decided the best control strategy would be to oil eggs and shoot adults at maximum feasible levels: oiling 70 % of ground nests and shooting 2000 adults / year. Although oiling 50 % of ground nests was apparently predicted to meet control program goals, 70 % was selected because "the only risk in doing so was to reach our goal more rapidly."

Effectiveness of the control program was more dramatic than the model predicted, and led to the decision to stop shooting after four years, rather than the five that had been planned

(see St. Lawrence Estuary, Québec in PART V. MANAGEMENT OPTIONS). Why might reality have produced results so different from those predicted by the model? Given lack of data on DCCO life-table parameters and lack of model sensitivity analyses, it is difficult to answer this question. All parameters were based on best guesses and tweaking of the initial model to reproduce a historical pattern consisting of two points. Important assumptions were also made that may not be valid. Analyzing the control program, Bédard et al. (1999) state that unequal sex ratios in DCCOs shot (203 males: 100 females) “probably accounted for the faster than predicted drop in numbers.”

The model predicted that control would not be successful without shooting adults, a strategy those involved wanted to avoid because of negative publicity it was expected to draw. (After completion of the control program, it was reported that, despite attempts to manage public opinion, “press coverage was acerbic and negative.”) Given that the selected control program was considerably more effective than predicted, it is possible that the goal of reducing the population to 10,000 pairs might have been achieved without shooting adults. In that case, the model would not be considered successful at predicting the most appropriate control strategy under current social conditions. However, even in hindsight it is not possible to make definitive judgments without more data.

Lake Champlain, Vermont

Fowle (1997) constructed a deterministic, age-structured, exponential model of DCCOs on Lake Champlain, Vermont with the stated goals of 1) comparing the model’s predicted growth rates to observed DCCO growth rates on Lake Champlain, and 2) evaluating proposed DCCO control methods. Sensitivity analyses were performed on two of the model’s variables (mortality and reproduction rates), but the importance of other unknowns / assumptions remained untested. The model’s predicted rate of increase, r , at stable-age distribution in the absence of control was found to be 0.21, which is reported as comparable to observed growth rates. Of the three control strategies modeled, reducing adult survival by 75 % was found to be the most effective strategy at lowering r . Preventing reproduction on all islands except the Four Brothers and Shad Islands (described as the control technique “with the least amount of disturbance”) was the next most effective technique, while reducing adult survival by 10 % reduced r by only a small amount (see below).

This modeling effort appears to be based on two separate models: an age-structured exponential model and a basic exponential growth model. The age-structured model used life table parameters (both known and unknown for DCCOs) to estimate r without control and under various control strategies. The basic exponential model was then used to project 1996 population estimates into the future given various r values. Effectiveness of control strategies was measured by how close each strategy’s predicted r value came to zero.

The age-structured model consisted of 15 age classes (0 – 14). A reproductive rate of 2.54, based on fledging rates observed on Young Island, Lake Champlain (1995 – 1996), was applied to all age classes 3 – 14 (i.e. no age-specific fecundity). All adults were assumed to breed every year (despite control efforts to prevent colonization on some islands on Lake Champlain). Age at first breeding was assumed to be three years. Age-specific survival rates (58 % for age 0, 75 % for 1 and 2, and 80 % for 3 and above) were not based on DCCO data. Rather, they were taken from a model by Madenjian and Gabrey (1995) who used Herring Gull mortality rates reported by Paynter (1947) for DCCO hatchlings (here age 0) and non-breeding birds (ages 1 and 2), and Cairns et al. (1991) for breeding birds (ages 3 and above). The 80 % survival rate of

Cairns et al. (1991) is reported as a “typical literature value” for the general class of inshore seabirds. Because population age distribution was unknown, r was calculated from the model when it reached a stable age distribution. (Age-structured exponential models reach a point at which r and the proportion of individuals in each age class no longer change from year to year; these proportions are the stable age distribution.)

Sensitivity of the model was tested by varying survival and fledging rates by 10 % and observing effects on r . These tests revealed that the model was more sensitive to variation in survival than reproduction: 10 % change in survival rates led to 40 % change in r , whereas 10 % change in fledging rates resulted in 10 % change in r . Thus, of these two parameters, the model was most sensitive to the one that was unknown and had been estimated from other species, and least sensitive to changes in the variable that was calculated directly from the population in question. Variation in other unknowns such as age at first breeding and proportion of non-breeding adults was not tested.

Three control strategies were modeled. The effects of oiling all eggs on Young Island (see Lake Champlain, Vermont (USFWS 1999a) in PART V. MANAGEMENT OPTIONS) was simulated by reducing reproduction (in an unexplained manner) to simulate reproduction occurring only on the Four Brothers and Shad Islands. To simulate two levels of lethal control of adults, adult survival rate was reduced by 10 % and 75 %. The simulated egg oiling resulted in an r of 0.06 at stable age distribution. Reducing survival of adults by 10 % resulted in a r at stable age distribution of 0.18, while reducing adult survival by 75 % resulted in an r of 0.02. The level of lethal control necessary to reduce adult survival rates by 75 % is reported as equal to killing over 12 % of the breeding population.

Using the above r values to predict future population growth makes the assumption that the population is currently at a stable age distribution. This assumption may not be valid. Fowle (1997) notes that the model predicted birds of age 1 and 2 would comprise 34 % of the population at stable age distribution, whereas observations of foraging flocks found these age classes comprising only 10 %. This discrepancy could have resulted from several scenarios. The Lake Champlain DCCO population could be at a stable age distribution that is different from that predicted by the model because the reproduction and mortality parameters in the model are incorrect. Or, the model’s predictions could be correct, but the population may not be at a stable age distribution. Or, the model could be correct and the population could be at the predicted stable age distribution, but observations of the population’s age classes may be incorrect. Given available data, it is not possible to determine if any of these scenarios explains the discrepancy, however the accuracy of the sub-adult to adult ratio estimate may be questionable. It appears that authors assumed the sub-adult : adult ratio in foraging flocks is equal to the same ratio in the entire population. This may not be the case. Because breeding adults must provide food for both themselves and their young, energy requirements for breeding adults are greater than for non-breeders; therefore, foraging effort of breeding adults should be higher than that of non-breeders. (The bioenergetics model in this study predicted a higher food intake for breeding cormorants compared to non-breeding cormorants.) In growing populations, a large proportion of non-breeders generally will be sub-adults. (The model assumed all non-breeders were sub-adults.) If foraging efficiency of breeders is similar to that of non-breeders, and if non-foraging birds do not associate with foraging flocks, then non-breeders would be expected to appear in foraging flocks less frequently and for shorter durations than breeders. Therefore, the sub-adult : adult ratio in foraging flocks (as calculated from observations throughout the nesting season), may be lower than this ratio in the population as a whole.

As with the St. Lawrence Estuary, Québec, model (above), this model assumes that mortality to adults through lethal control will be equal for both sexes. Because unequal mortality to males and females during the St. Lawrence Estuary control program may have contributed to the higher effectiveness of the control program than predicted (Bédard et al. 1999), the effectiveness of lethal control on r values reported here should be interpreted with caution. Nevertheless, the sensitivity analyses performed on this model are instructive and highlight the need for better age-specific DCCO mortality rates.

DETERMINISTIC LOGISTIC MODELS

Basic Logistic Growth Models

Logistic models attempt to incorporate the density-dependent factors that limit population growth. In the basic logistic growth model, all factors are combined into one number, the population's carrying capacity, or K . As the population grows, its growth rate, r , is reduced such that as the population approaches its carrying capacity $r \rightarrow 0$ and growth eventually stops. These models make the assumptions that a fixed carrying capacity exists and that the population will increase to this level in a sigmoidal fashion. Mainly, these models are used to estimate what a growing population's carrying capacity will be, and not to predict the population's trajectory per se. This goal is accomplished by fitting population data to various logistic growth curves using regression analyses, and then analyzing how close the fit is.

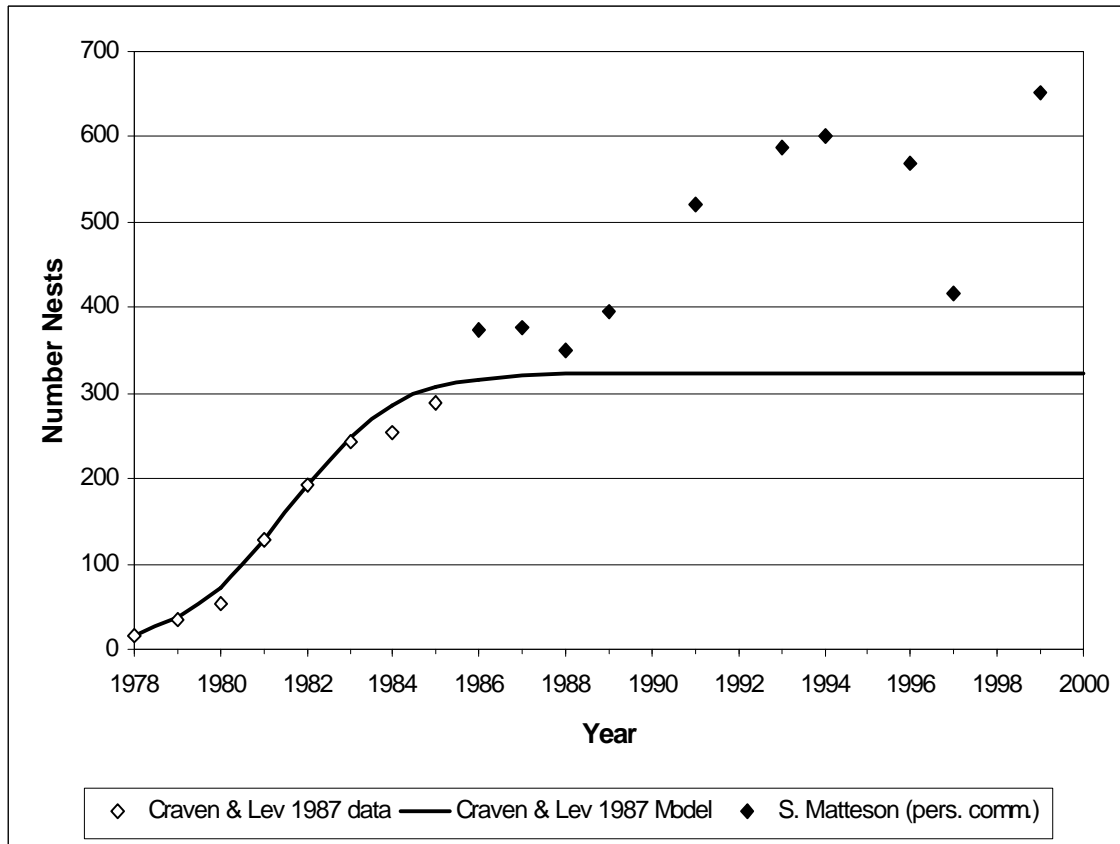
The only application of the basic logistic growth model to DCCO data appears to be Craven and Lev (1987), who apply this model and nest count data for Gull Island, Lake Superior (1978-85) to predict island carrying capacity. They fit their data to the logistic equation

$$N_t = \frac{k}{1 + Ae^{-rmt}}$$

and found that the best fit curve was described when $k = 324$, $A = 18.041$ and $r_m = 0.818$. Thus, their model predicts a carrying capacity of 324 nests. However, nest count data after 1985 (S. Matteson, pers. comm.) reveal island carrying capacity is much higher, possibly yet to be reached (Figure 92). This example reveals that logistic curve fitting may be sensitive to recent population counts (e.g., 1984 and 1985 in Figure 92), and that carrying capacity predictions for populations just beginning to show signs of slowed growth are subject to great uncertainty.

Suter (1995) uses the basic logistic curve to analyze growth trends in the GRCO population wintering in Switzerland from 1967 to 1992. Population data were fit to three different models (logistic, power and exponential) and statistical analyses were applied to each to determine which gave the best fit. Although the correlation coefficients for all three were high ($r = 0.996$, $r = 0.986$ and $r = 0.977$, respectively) the power and exponential curves "were inferior to the logistic model, as shown by the high autocorrelation of the residuals (logistic: $r = 0.079$, n.s.; power: $r = 0.626$, $p < 0.001$; exponential: $r = 0.715$, $p < 0.001$)." Two carrying capacity values were predicted by the model depending on whether a low 1991 count was included in the analyses: 8800 cormorants with 1991 data and 9400 without. Although it would be instructive to compare these predictions with data collected after 1992, we have not acquired these data. Neither of the above logistic curve fitting examples gave confidence intervals around predictions carrying capacities.

Figure 92. Nesting population change on Gull Island, Lake Superior and logistic model based on 1978 – 1985 nest counts.



Age-Structured Logistic Growth Models

If the effect of density-dependence on various life history parameters such as survival and reproduction are known, it is possible to incorporate these interactions into age- and / or sex-structured models. Such models could be useful for predicting a population carrying capacity in the absence of trend data. These models could also be valuable for investigating effects of density dependence on attempts to control population levels. For example, density-dependent effects on reproduction may increase per capita reproductive output at low population levels, thereby increasing effort necessary to manipulate populations to lower and lower levels in a nonlinear fashion. An understanding of such interactions would be valuable for deciding what levels of population control might be economically feasible.

Data on effects of population density on life table parameters are unavailable for DCCOs. For example, Ludwig and Summer (1995) discuss ways in which human factors, disease, parasites, availability of nesting habitat and availability of food may limit DCCO populations, and they provide anecdotal evidence that some of these factors may be acting on the Great Lakes population, but they provide no quantitative data that could be used to model these effects.

GREAT CORMORANTS IN EUROPE

Bregnballe et al. (1997) constructed a deterministic, age structured model with density dependence for the European GRCO *sinensis* population to investigate the ability of various management techniques to manipulate population levels. The model was created by modifying a preexisting model originally constructed for oystercatchers with parameters derived from GRCO research. The parameters in which density dependence was incorporated were the proportion of non-breeders, chick mortality and winter survival. Mortality was density independent for eggs as well as for juveniles and adults during the spring, summer and autumn. Clutch size was not age structured or density dependent. Parameters such as initial population size, age class distribution and area of breeding and wintering habitat were mentioned but not discussed.

Similar to DCCO modeling efforts, many of the parameters necessary for constructing this model were either variable, poorly known, or unknown; selection of parameter values consisted largely of educated guesses. The model was calibrated by adjusting the strength of density dependence and availability of resources “until the unmanaged population behaved in a way that was within the range of what [the authors] found would be likely to occur,” and so that the population stabilized “before reaching an unrealistically large size” (Bregnballe et al. 1997:77). In other words, the model was constrained a priori by assumptions of overall population limitations.

Because of the level of uncertainty in the parameters, sensitivity analyses were run to determine how this uncertainty affected model output. Variables were changed individually and observations were made of subsequent changes in model behavior. Assumptions about density dependence were found to be very important in determining model output, with 1) amount of available habitat and strength of competition within breeding habitat affecting the proportion of adults breeding and 2) fledging rate and survival of young birds affecting final population size. Age of first breeding was found to affect size of the non-breeding population, rate of approach to equilibrium, and final population size.

Four different management scenarios were modeled independently, and then in various combinations. Because of low level of confidence in many model parameters, little confidence was held in the predicted final population sizes. Effectiveness of management techniques was measured by comparing extent by which these techniques lowered the estimated equilibrium levels when compared to model predictions without any management of the cormorants. The four management scenarios were: preventing establishment of new colonies, cutting nesting trees and disturbing breeding birds, reducing reproductive output, and introducing a hunting season in Europe.

Considerable detail is given in discussing the results of each management simulation, both in quantitative and qualitative terms. Overall, it was concluded that each management scenario could be effective at holding the entire *sinensis* population below its natural carrying capacity. The authors “conclude that shooting adult cormorants would be more efficient than nest destruction and egg culling partly because of compensatory mechanisms and partly because of the direct and indirect effects of hunting on mortality. Preventing new colonies from being established could be efficient when and where colonisation of areas of intermediate or high quality could be hindered.” All combinations of management techniques were found to be more effective than the techniques applied separately. The authors end with the following caveat:

Although the model predicts that all the above management measures would lead to stabilisation at a lower population level than if no actions were taken, a best professional judgment suggests that a substantial reduction in population size does not necessarily lead to a substantial reduction in the number of cormorants foraging in so-called problem areas. Furthermore, economic losses would not necessarily decline proportionately with a decline in the number of cormorants foraging in a problem area.

CONCLUSIONS

Review of these studies indicates that prediction of future DCCO population trends and analysis of control methods is hampered by a lack of age- and sex-specific data for this species. Even when proper models are selected for available data and sensitivity analyses are performed on assumptions and unknowns, the questions that can be answered by these models are limited to the more basic components of cormorant population dynamics. Until better DCCO life table data are available, sensitivity analyses of assumptions and unknowns will be very important for assessing confidence levels in population model predictions.

PART VII. RESEARCH NEEDS

Research needs have been identified by multiple researchers and through our DCCO Status Assessment. Additionally, an interagency meeting was convened March, 2000, in Sandusky, Ohio, to discuss Research Needs for North American Cormorant Management, and ideas from this meeting have been incorporated. In our assessment, we have specifically focused on research necessary for managing and resolving human/cormorant conflicts, and have attempted to group research needs in order of priority. With the exception of number 25, all of the needs presented below were discussed at the Sandusky meeting, and each was ranked as a 1, 2 or 3 in terms of importance for management of the DCCO problem. A “1” designated the highest importance, while a “3” designated not important for management. Number 25 was later suggested by John Trapp, and ranked by a few individuals present at the Sandusky meeting and Jeremy Hatch.

Research needs were identified and ranked in the following categories:

- Distribution
- Demographics
- Diet
- Impacts to Fisheries and Aquaculture
- Impacts to Flora and Fauna
- Management Techniques

DISTRIBUTION INFORMATION

- 1) Information is needed on proportion of total population utilizing aquaculture facilities, and on how and where winter numbers are distributed (winter distribution as it relates to “local features”; Trapp et al. 1995). (2)
- 2) Information is needed on distribution, numbers and habitat use of non-breeding birds during the breeding season. (1)
- 3) Better delineation of interior, southern and northeast Atlantic Coast breeding populations; boundaries are becoming blurred. Genetic studies would be useful to examine distinctiveness and extent of gene flow between populations (Hatch and Weseloh 1999). (3)
- 4) Better utilization of historic distribution information to help distinguish range expansion from re-colonization. (3)

DEMOGRAPHIC INFORMATION

- 5) To model management scenarios and effectiveness, to predict future trends, and to determine factors that may limit populations, more information is needed on population/life table parameters (Brugger 1995; Erwin 1995; Nisbet 1995; Hatch and Weseloh 1999), such as:

- age of 1st breeding (1)
- reproductive output (LRS and age-specific fecundity) (1)
- frequency of breeding (annual?) (3)
- mortality sources (3)
- age-specific survival/mortality (1)
- emigration and immigration: local (1); regional (2)
- philopatry in DCCOs (important for local population control effectiveness): local (1); regional (2)

Long term studies (banding with hardmetal and field-readable bands¹, etc.) will be necessary to acquire these data. Until these DCCO data are available, known parameters for European GRCO could potentially be used for DCCO modeling. Additionally, effectiveness of various management scenarios has been modeled for European GRCOs (Bregnballe et al. 1997).

- 6) Better estimates of numbers of breeding birds within each zone (e.g. lack of surveys, sampling inconsistencies). To produce population estimates with high confidence, both improved geographic coverage and a standardized sampling method are necessary. (1)
- 7) Better estimates of numbers of wintering birds in specific locations (1) and overall (3).
- 8) Studies to ascertain how population processes are affected by changes in density, as in the European Great Cormorant population (e.g., Van Eerden and Gregersen 1995; Frederiksen and Bregnballe in press). (1)²
- 9) Information on factors influencing colony size, formation and movement [e.g., dispersal behavior as it relates to selection of new areas for colonization (Bregnballe and Gregersen 1997)]. (2.5)

DIET

- 10) Experimental feeding trials should be undertaken to assess the biases associated with pellet and stomach content analyses. Trials could be used to determine the most appropriate (species-specific) key bones and other hard parts to be used for analysis. Then attempts could be made to quantify size- or species-related differential recovery (Carss et al. 1997). (2)³
- 11) Need standardized methods for analysis of pellets and stomach contents to make studies more comparable (Carss et al. 1997). (2)³
- 12) Better information on rate of pellet formation and influencing factors (Carss et al. 1997). (2)³

IMPACTS TO FISHERIES AND AQUACULTURE

- 13) Assess mortality due to DCCOs relative to other mortality factors at the local level (Price and Nickum 1995; Trapp et al. 1995; Erwin 1995). Better estimates of numbers and densities of fish populations will be required to do this. (1)

- 14) Better information is needed to explain mechanisms within fish populations that buffer effects of DCCO predation (i.e. compensatory vs. additive mortality due to DCCO's) (Krohn et al. 1995; Erwin 1995; Nisbet 1995; Carss et al. 1997). Care should be taken to investigate whether different life-stages or species complexes of fish are more or less vulnerable to cormorant predation (i.e. differential effects on young-of-year vs. adult fish; forage vs. predator fish species). (1)
- 15) More information is needed on DCCO foraging behavior and predator-prey interactions (Carss et al. 1997; Hatch and Weseloh 1999). (2)
- 16) Information is needed on intra-specific competition, dynamics between individual birds and possible density dependent effects on foraging choices, behavior and patterns (Nisbet 1995; Bregnballe pers. comm.). (3)
- 17) Studies to examine the potential beneficial role of the DCCO at aquaculture ponds (e.g., remove dead and diseased fish; reduce densities of uncontrolled fish; eliminate stocked fish fry predators; indicator of undesired conditions at ponds; Jackson and Jackson 1995). See Ashkenzai and Yom-Tov (1996) for similar information on the role of herons at fish ponds in Israel. (2)

IMPACTS TO FLORA AND FAUNA

- 18) Research should be conducted to quantify impacts to vegetation and to other waterbirds. Sites where DCCO induced vegetation changes may harm rare plant communities or animals should be identified, and impacts to survival / reproduction of avian species should be examined. (1)
- 19) Research is needed to determine how Newcastle disease may be transmitted from DCCOs to commercial poultry and to identify risk factors involved. Research is also needed to determine whether DCCOs transmit Newcastle disease to other wild birds (Kuiken 1999). (3)

MANAGEMENT TECHNIQUES

- 20) States where the Depredation Order is in effect should closely monitor DCCO numbers to determine impacts of and to DCCOs. The number of birds taken under the Depredation Order and effectiveness of the Order in terms of buffering DCCO impacts should be assessed. A comprehensive study of birds killed under the Depredation Order to determine age and sex of birds depredating at aquaculture facilities, relative to the population as a whole, should be conducted. (1)
- 21) More research is needed on how to make control measures more cost effective. Many techniques are poorly tested and should be investigated more thoroughly. (1)
- 22) Studies to address the human dimensions of DCCO conflicts and possible solutions through education should be undertaken. (1)

- 23) Encourage communication between fisheries biologists and ornithologists to determine how piscivorous birds can best be incorporated into fisheries management objectives (e.g., stocking sportfish species or sizes less vulnerable to DCCO predation; adjust stocking rates to accommodate some degree of predation; in areas where there is conflict, create alternative habitat to minimize potential competition for fish resources; change the time of stocking to avoid cormorant predation; encourage monitoring of return rates of adult fish or creel fish to determine if cormorant predation has a significant impact on overall fish returns) (Ottenbacher et al. 1994). (1)
- 24) For species that may be taken accidentally in lethal control efforts, such as GRCO, Anhinga and Neotropic Cormorants, more research is needed about how losses may impact these species on a regional or population level. For example, observations at ponds to determine proportions of each species present and taken may provide useful data on losses and impacts. (3)

ECONOMIC IMPACTS

- 25) Research on economic impacts, especially as it relates to the sport fishing industry, tourism, etc., should be conducted. (1)

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- 1 Importance of hardmetal bands should be emphasized; cormorants outlive bands made of aluminum, so that data for birds more than a few years old are suspect. Use of appropriate bands is essential for accurate measures of adult survival. Additionally, large bands that are readable through a telescope, such as those used in Denmark and elsewhere, will be necessary for measuring survival and movements, and can provide important information on the dynamics of colony formation (J. Hatch, pers. comm.).
 - 2 After discussion with Jeremy Hatch, number 8 was re-written and as re-stated is more important than in its earlier form, so its rank was changed from "1.5" to "1".
 - 3 We originally listed number 11 before number 10. Additionally, numbers 11 and 12 were originally ranked as "1" while number 10 was ranked "2". However, after discussion with D. Carss and J. Hatch, we made the above changes in order and rank for the following reasons: Ranking number 10 as a "2" while numbers 11 and 12 were ranked as "1" implies that the pellet method yields representative samples for establishing daily food intake, but this method is unsuitable for estimates of DFI (see Diet Assessment Methods: Advantages and Shortcomings); Pellet analyses are not a priority research need for management of cormorants, since potentially more reliable methods (e.g., stomach contents analysis) may be used to assess cormorant diet. If biases associated with pellet analyses remain unknown, numbers 11 and 12 aren't meaningful.

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APPENDIX I. PERCEIVED IMPACTS ASSOCIATED WITH DOUBLE-CRESTED CORMORANTS

To determine if states and provinces perceive DCCOs as important impacts to biological resources and human interests, we included questions regarding perceptions of DCCO impacts in our DCCO survey. Recipients of the survey were asked to provide information on impacts to the following resources: commercial and or sport fishes, aquaculture, vegetation, other vertebrate species (birds, mammals, herps, etc.), private property, and water quality. We also requested information on disease transmission between cormorants and domestic or wild birds, and impacts to public health. Table 29, Table 30 and Table 31 summarize perceived impacts to each resource by region. We also developed a survey for fisheries biologists to obtain information on impacts to fisheries from the fisheries community perspective; these data are summarized in Table 32 and Table 33.

The range of knowledge about DCCOs and their possible impacts varied greatly among survey recipients. Additionally, recipients were simply asked to give their own or their agency's opinion on the questions being asked. Therefore, it is important to note that the information in this section is in no way intended to document impacts. The data presented here are mainly qualitative, and are included only to provide a broader perspective on attitudes of wildlife and fisheries agencies and biologists towards cormorant issues.

SURVEY RESPONSE

Overall, we obtained a good response to our survey requesting information on DCCO populations, impacts and management. We sent surveys to 61 individuals, and of these, about 80 % (49 individuals) responded by either completing the survey or by contributing partial data. In most states and provinces, one individual was contacted to complete the survey. However, in some cases, two or more individuals were sent surveys. This was done when we determined through telephone contacts that more than one person would be necessary to supply all requested information; or because DCCO problems were documented in an area and specific individuals could contribute important information and or a wider perspective. Data are summarized in Table 29, Table 30, Table 31 and below.

For our survey requesting information from fisheries biologists, we sent surveys to 35 recipients, and of these about 66 % (23 individuals) completed surveys. This information is summarized in Table 32, Table 33 and below. Additionally, some individuals completing our DCCO survey consulted with fisheries biologists to complete sections regarding impacts to fisheries and aquaculture.

Impacts to commercial / sport fish (reported by wildlife biologists, etc.)

- 1) *Respondents were asked to rank the importance (minor, moderate or major), in the opinion of their agency, of Double-crested Cormorant predation in limiting economically important (game and / or commercial) fish species in their state / province.*

The majority of states and provinces reported that DCCO predation was perceived to be of minor importance to sport / commercial fish populations. Only two states and one province

reported that DCCO predation was perceived to be of major importance in limiting fish populations.

Southern U.S.: Two states reported that DCCO predation was perceived to be of major importance in limiting fish populations in the southern U.S., Texas and Arkansas. However in Arkansas, two persons completed surveys, a USDA/WS District Supervisor, who reported DCCO predation was perceived to be of major importance, and a Nongame / Endangered Wildlife Program Coordinator, who reported predation was perceived to be of moderate importance. Three states reported that DCCO predation was perceived as moderately important. In addition to Arkansas, Louisiana had two individuals complete surveys, a USDA/WS biologist, who reported that impacts to fish were unknown, and a nongame biologist who reported that DCCO predation was perceived as moderately important. Respondents from the other seven states in the region reported that DCCO predation was perceived to be of minor importance. Oklahoma reported that importance of DCCO predation on fish populations was unknown.

U.S. and Canadian Interior: All survey respondents reported that DCCO predation was perceived to be of minor importance, with the exception of Manitoba, where predation was perceived to be of major importance.

North Atlantic: Twelve respondents from eight states and two provinces reported that DCCO predation was perceived to be of minor or no importance; five respondents from six states reported that it was perceived to be of moderate importance (the same individual responded for CT, MA and RI). There were two respondents from Rhode Island; the USDA/WS biologist reported impacts to be of moderate importance while the state biologist reported that importance of DCCO predation on fish populations was not known. Though a survey was not completed for New Brunswick, Korfanty et al. (1997) reported that Double-crested Cormorants are not viewed as a problem in this province with respect to impacts on commercial fisheries.

2) *Respondents were asked if there were any current studies underway to assess impacts in their state / province.*

Current studies were reported in each region in a total of 10 states and provinces. These studies are presented in Appendix II. Current Research, Table 34.

Impacts to commercial / sport fish (reported by fisheries biologists)

1) *Respondents were asked if commercially valuable fish species had declined in their area, and if so, to list the species in decline and factors thought responsible for the declines.*

Declines were reported in 16 states and provinces. Fish species reported in decline and factors thought responsible are listed in Table 33. Overfishing and habitat degradation (loss of spawning habitat, pollution, dams, poor water quality, phosphorus abatement, etc.), were the most frequently reported causes of decline. Other factors reported included exotics (zebra mussels, sea lamprey, etc.), predation (by other fish and DCCOs in 5 states), white perch invasion, global and local environmental change, recruitment failures, lack of zooplankton, weather, temperature, and the natural cycle.

- 2) *Respondents were asked if fish declines could be linked to Double-crested Cormorant increases, and if so, to please supply data to support this link.*

Five individuals from five states reported declines in fish populations that they believed were linked to DCCO predation. Four of the five areas where links to DCCOs were reported were in the Great Lakes; the other was in the Northeast. Maine and Michigan reported only anecdotal data linking DCCOs to fish declines. In Vermont, declines in yellow perch were correlated with DCCO increases; trends in creel survey estimates versus growth of Lake Champlain cormorant populations from 1990 through 1997 are being analyzed, but at the time of this writing no concrete data were available to support this link. Wisconsin provided a report that included rough estimates of the number of yellow perch consumed by DCCOs during the 1983 breeding season at Willow and Cat islands. The report also included estimates of the potential amount of perch in pounds lost to DCCOs, but did not incorporate data on the total number of perch present. Since that report was written, a model has been developed that has allowed the reconstruction of annual biomass estimates for yellow perch in southern Green Bay, and the amount consumed by DCCOs, estimated at 5.96 % in 1983. New York also provided recent reports that included estimates of fish abundance and percentages consumed by DCCOs (VanDeValk et al. 1999; NYSDEC 1999 1999), and felt that significant impacts to smallmouth bass and possibly to walleye were occurring (See PART III: DIET, studies assessing impacts to sport and or commercial fish populations, for limitations of these type of data in assessing relationship between fish population declines and DCCO predation).

- 3) *Respondents were asked if there was any information on the percentage of commercially important fish populations consumed by DCCOs, and to include any important reports that address this type of information for their state / province.*

Respondents from New York and Wisconsin provided us with reports or data (see #2 above) that provided information on the percentage of commercially important fish populations consumed by DCCOs. The respondent from Michigan referenced the NYSDEC's Final Report (1999), noted above, and a report by Diana et al. (1997). Data from the latter report was discussed and analyzed by Belyea et al. (1999), and summarized earlier in (PART III: DIET). Belyea et al. (1999) and Diana et al (1997) reported that DCCOs removed about 1 % of the legal sized yellow perch population in the Les Cheneaux Islands area, Michigan, and were not believed to be causing substantial declines in the population.

Impacts to Aquaculture

- 1) *Respondents were asked to rank the degree (minor, moderate, major) to which DCCO predation at aquaculture facilities was perceived to be a problem in their state / province.*

DCCO predation at aquaculture facilities was perceived as a major problem only in the southern U.S. and in Manitoba. In the Great Lakes, the northeastern U.S., and Canada, it was mostly perceived to be a minor problem. A total of five states, Illinois, Connecticut, Massachusetts, Rhode Island and Maine, reported it to be perceived as a moderate problem. In the northeastern states, both USDA/WS personnel and state agency wildlife biologists completed

surveys; USDA/WS personnel reported it perceived as a moderate problem while state agency biologists reported it perceived as a minor problem.

2) *Respondents were asked to estimate the economic loss due to DCCO predation at aquaculture facilities in their state / province.*

Only four states provided actual dollar amounts associated with DCCO predation at aquaculture facilities. In the southeastern U.S., LA, AL, and MS estimated dollar amounts in the millions. In the northeast, VA estimated losses of about \$15,000. In most states and provinces, the dollar amount of loss was unknown.

3) *Respondents were asked how many (most, some, few) aquaculturalists in their state had taken DCCOs under the provisions of the DCCO cormorant depredation order.*

Cormorants were taken around aquaculture facilities mainly in the southern U.S. We received replies of “most” only in Arkansas and Mississippi. Replies of “some” were given for Texas, Louisiana, Alabama, and Florida. In general, most states responded with few, none or unknown to this question.

In Canada, the DCCO is protected and managed under provincial law. Scare and kill permits have been issued to “some” aquaculturalists in Manitoba, and to “few” on Prince Edward Island. The other provinces we surveyed did not report issuing any permits.

Impacts to Vegetation

1) *We asked respondents if vegetation (categorized as herbaceous layers, trees, and rare or unique plant species / communities) had been affected by DCCOs in their state / province.*

Impacts to vegetation, mainly to trees, were reported in each zone, but most frequently in the northeastern U.S. and Canada. Eight northeastern states, CT, ME, MD, NH, NY, RI, VT, and VA, reported impacts either to trees (7), the herbaceous layer (5), or both (4). All five Canadian provinces reported impacts to trees, while Manitoba and Ontario also reported impacts to herbaceous layers. In the southern U.S., impacts to trees were reported in OK, AL, AR, FL, and NC; impacts to herbaceous layers were also reported in NC. In the Great Lakes, impacts to both herbaceous layers and trees were reported in IA, OH, MI and WI. No states or provinces reported any known impacts to rare plant species or communities, with the exception of Ontario, which reported that the unique Carolinian habitat on East Sister Island in Lake Erie might be affected by DCCO nesting. Ontario plans to assess the site this year if funds are available (J. Harcus, pers. comm.).

Impacts to trees were reported to be due mainly to guano deposition, and resulted in tree die off at breeding colonies and roost sites. Arkansas reported that guano deposition at roost sites stunted growth of buds and leaves in spring; Mississippi reported a similar concern over impacts to spring tree growth at roost sites. Other reported impacts included “tree pruning” by DCCOs when building their nests; on some islands in Iowa, all small branches were removed from trees for nesting material.

Impacts to the herbaceous layer were also reported due to guano deposition, and often this layer was reduced or eliminated from the colony site. In some areas (MB, VT), the only herbaceous vegetation remaining around cormorant breeding colonies were nitrogen tolerant

species, such as stinging nettles and thistles. Michigan reported that in some breeding colonies, diverse native island plants were replaced with “barnyard” species. Virginia reported that destruction of the herbaceous layer made low islands subject to flooding.

Impacts to avian species

1) Respondents were asked if the recent increase in DCCOs had impacted avian species in their state / province, and if there were such impacts, to describe them..

Impacts to avian species were reported in each zone, and were reported to occur mainly through habitat degradation (effect of DCCO guano on trees and ground cover) and competition for nest sites. Impacts to other colonial waterbirds, particularly herons and egrets, were reported most frequently. In the Great Lakes, Ohio reported that DCCOs might displace Great Blue and Black-crowned Night Herons and Great Egrets on West Sister Island. Michigan reported that DCCOs may have caused the relocation of Great Blue and Black-crowned Night Herons and Caspian Terns on some islands. Illinois reported that Black-crowned Night Herons may have been displaced from one rookery by DCCOs, but that Great Blue Herons had also moved in and thus may have caused Black-crowned Night Heron dispersal. Wisconsin reported that Herring Gulls were displaced from some areas on islands, but were not eliminated or forced out. Iowa reported that Canada Geese and Mallards nested on islands on the Mississippi River that were later denuded by DCCOs, resulting in increased predation and abandonment by these species.

In the southern U.S., Arkansas reported that DCCOs had excluded Cattle Egrets and Little Blue Herons from nesting areas at a rookery in Grassy Lake, and that egrets, herons, and other waterbirds had been replaced by DCCOs at a swamp in the southwestern portion of the state. Mississippi reported speculation that large concentrations of night roosting DCCOs may displace wintering waterfowl from roosting, feeding and loafing sites.

In the northeastern U.S., several states reported impacts or potential impacts. New York reported that DCCOs nesting at Little Galloo Is., Lake Ontario, displaced Black-crowned Night Herons. Vermont reported impacts to Black-crowned Night Heron, Cattle Egret, Gadwall, Mallard and American Black Duck. All of these species previously nested on Young Island, but vegetation was impacted by DCCO guano deposition. Most of the island is now denuded, and characterized by DCCOs and gull species; none of the species mentioned above currently nests on the island. Potential or possible impacts were reported in ME, MD and MA. Massachusetts reported that “it is reasonable to conclude that there has been a small amount of displacement of Herring Gulls in their nesting areas, but this has not been quantified.” Maine reported that DCCOs nesting on islands with Common Eiders reduced the available habitat in the vicinity of the DCCO colony, but that these changes are temporary and not limiting to other colonial nesters in most cases. Maryland reported that in the late 1990s, DCCOs moved into a heronry where they may eventually exclude herons because of guano impacts. The island also supports Common Terns, Herring Gulls and Great Black-backed Gulls, which could be impacted in 5-10 years.

In Canada, Manitoba reported that DCCOs trample Common Tern colonies, and kill trees used by Great Blue and Black-crowned Night Herons. Additionally, they compete for habitat with Ring-billed and California Gulls, Caspian Terns and American White Pelicans. Prince Edward Island reported that DCCOs probably have an impact on Black-crowned Night Herons, crowding out herons and killing nesting trees at a greater rate than herons would have done.

Ontario simply reported impacts to Black-crowned Night Herons, but didn't explain what these impacts were.

A few respondents also reported positive impacts to avian species associated with DCCO nesting habits. Bill Koonz reported that DCCOs in Manitoba create habitat for themselves and for other species, such as plovers, terns, avocets, etc., by setting back vegetation and providing open areas. Similarly, David Allen noted that the tendency of DCCOs to reduce herbaceous vegetation is probably beneficial for some of the other colonial nesters in North Carolina that require bare sand for nesting.

Another important observation was that illegal cormorant control in Manitoba appeared to have substantial impacts for colonial waterbird breeding ecology on Lake Winnipegosis. Bill Koonz reported that during the 1999 colonial waterbird survey, most traditional cormorant colonies along the main travel lanes of Lake Winnipegosis had been disturbed, and cormorants had abandoned these sites. Ring-billed and California gulls, and Caspian and Common Terns also nest at these sites, but their nests had not been disturbed. Cormorants appeared to have a buffering effect within this community, previously nesting among and between these species. However, without DCCOs to buffer nesting cells between the gulls and terns, these species nest together, and considerable predation on nests occurs when these islands are visited by humans (where the nesting cells overlap, all the nests of Common Terns and Ring-billed Gulls are quickly destroyed). Additionally, when cormorants and pelicans are not on these islands, the islands begin to re-vegetate and the spaces used by the remaining species overlap. The result is under utilization of the available nesting space, destruction of nests, young and eggs by competing species, and a gradual loss of habitat on the colony due to vegetation encroachment. A number of islands on Lake Winnipegosis appeared to demonstrate this phenomenon.

Impacts to other fauna

- 1) *Respondents were asked if the recent increase in DCCOs impacted any other terrestrial fauna (mammals, herps, etc) in their state / province, and if so, to describe any impacts.*

Iowa was the only state to report impacts. Nesting DCCOs were believed to have reduced the diverse fauna that formerly occupied five islands on the Mississippi River, but no additional information was provided.

Damage to Private Property

- 1) *Respondents were asked to provide information on damages to private property by DCCOs (type of damage, estimated dollar loss, etc.).*

Only four states (SC, CT, ME and MA) and two Canadian provinces (MB and QC) reported damages to private property. South Carolina reported that damages to private property were minor and that affected land had not been reduced in value. Connecticut reported that a landowner complained about damages to expensive pinion oak trees and he actively discouraged DCCOs from nesting on his land. All trees survived and the affected land was not reduced in value. The Maine State Director of USDA/WS reported that he receives several calls a year about DCCOs eating trout out of private ponds. Loss varies from \$1000-25,000 at trout ponds and bait operations and may result in reduced land value. The Massachusetts State Director of USDA/WS reported that she receives 2-3 calls per year of DCCOs roosting on docks and boats

and making a mess with their droppings. She also reported that non-nesting DCCOs roost on power lines in Orleans, MA, and damage the herring run, pond water quality and residences around the lake, which has resulted in landowners in this area being unable to sell their homes. Manitoba reported that fish stocks are impacted but had no more specific information. The respondent from Quebec referred us to a paper by Bédard et al. (1995) which reported damages to unique forest habitats on several privately owned islands in the St. Lawrence River Estuary. No information was provided on dollar amounts of damage, etc. (see Part IV. Impacts to Vegetation and Colonial Waterbirds, for further information on this study).

Disease Transmission

1) Respondents were asked if there were any documented cases of disease transmission between DCCOs and wild or domestic birds.

Respondents from Michigan reported documented cases of disease in DCCOs. One reported that there had been a few cases of Newcastle disease in the past but not recently and not in any large numbers. The other reported a continental wide outbreak of Newcastle disease that occurred in 1992, and affected many DCCOs (mostly chicks), some White Pelicans, and a few domestic turkey flocks. The outbreak occurred during the summer, affecting breeding birds from the Rocky Mountains to Quebec, and is thought to have been picked up on the wintering grounds (J. Ludwig, pers. comm.). (See Part IV. Impacts to Vegetation and Colonial Waterbirds for further information on Newcastle Disease).

Impacts to Water Quality

1) Respondents were asked if there were any concerns in their state / province over increasing DCCO numbers and possible impacts to public health due to water quality and disease transmission.

Seven states and one province expressed concerns over possible impacts to water quality. In the Great Lakes, one respondent from Michigan reported that the Center for Disease Control was interested in human influenza viruses being carried by DCCOs and possibly transmitted through the water, but this has not been investigated and no additional information was provided. Ontario reported concern over the possibility that Newcastle Disease could be spread to domestic poultry. In the southeast, respondents from Alabama and Arkansas expressed concern over transfer of disease and parasites in aquaculture ponds, and that waste products from DCCOs may deplete oxygen and lower water quality. South Carolina reported that concerns about DCCO impacts on public health via water quality or disease transmission were slight but increasing. In the northeast, respondents from Maine reported concerns over DCCO and gull guano at lakes and ponds. The Massachusetts State Director of USDA/WS completed a survey for MA, CT and RI and reported that there was concern but did not specify in which states or the reasons for concern.

Impacts to Public Health

1) Respondents were asked if any instances of public health problems had been documented..

One Massachusetts respondent reported public health problems associated with DCCOs in Orleans, where a resident was reported to have asthma and allergies caused by odor of DCCO guano. Whether this was properly documented is not known.

Table 29. Impacts in Zone 3 (U.S. and Canadian Interior)

l = minor, m = moderate, h = major, p = possible, n = none, y = yes, u = unknown (states with > 1 response had > 1 respondent)

Resource Impacted		IL	IN	IA	MB	MI	MO	OH	ONT	SK	VT	WI
Sport & Commercial Fish	Degree of impact	l	l	l	h	l, l	l	l	l	l	l	l
	Studies	n	n	n	u	n, u	n	y	n	y	n	n
Aquaculture	Degree of impact	m	l	l	h	l, l	l	l	l	l	l	l
	Economic loss	N/A	n	l	u	u, u	l	n/u	n	n	n	u
	Legal take	n	n	few	some	n, u	l	n	n	n	n	n
Vegetation	Herbaceous	n	n	h	h	l, h	n	y	u	y	h	y
	Trees	n	n	m	h	l, h	n	y	l	y	h	y
	Rare/unique	n	n	n	u	n, u	n	p	u	n	n	u
Other species	Avian	?	n	y	y	y, y	n	y	n	y	y	y
	Nonavian	n	n	y	n	n, n	n	n	n	n	u	u
Private property		n	n	u	y	u, u	n	n	u	n	n	n
Disease transmission		n	n	n	u	y, *	n	n	n	n	n	n
Water quality		n	n	n	u	n, ?	n	n	y	n	n	n
Public health		n	n	n	u	n, u	n	n	n	n	n	n

* see text

Table 30. Impacts in Zone 4 (Southern U.S.)

l = minor, m = moderate, h = major, p = possible, n = none, y = yes, u = unknown (states with > 1 response had > 1 respondent)

Resource Impacted		AL	AR	FL	GA	KY	LA	MS	NC	OK	SC	TN	TX
Sport & Commercial Fish	Degree of impact	m	h, m	l	l	l	u, m	l	u	l	l	h	l
	studies	y	n, n	n	n	u	u, y	n	y	n	n	n	n
Aquaculture	Degree of impact	h	h, h	m	u	u	h, h	h	u	m	l	h	h
	Economic loss	Mil-lions	h, h	u	u	u	Mil-lions, u	5,000,000	u	u	u	u	u
	Legal take	some	most, most	s	u	u	some, u	most	u	u	u	some	u
Vegetation	Herbaceous	u	u, u	n	u	u	n, u	n	n	l	u	u	n
	Trees	h	y, l	l	u	u	n, u	u	y	l	n	u	n
	Rare/unique	u	u, u	n	u	u	n, u	n	n	n	u	u	n
Other species	Avian	n	y, y	n	n/u	n	n/u, n	?	u	n	u	u	n
	Nonavian	n	u, n	u	u	n	n/u, n	n	n	n	u	u	n
Private property		u	u, u	n	n	u	n, u	n	n	n	l	u	u
Disease transmission		n	n, n	n	n	n	n, n	n	n	n	n	u	n
Water quality		y	y, y	n	n	u	u, u	n	n	n	l	u	n
Public health		n	n, n	n	n	u	n, n	n	n	n	n	u	n

Table 31. Impacts in Zone 5 (North Atlantic)

l = minor, m = moderate, h = major, p = possible, n = none, y = yes,
 u = unknown (states with > 1 response had > 1 respondent)

Resource impacted		CT	DE	ME	MD	MA	NB	NH	NJ	NY	PA	PEI	QB	RI	VA	WV
Sport & Commercial Fish	Degree of impact	l, m	l	m, m	none	l, m	l	l	l	m	l, l	l	l	u, m	m	none
	Studies	y, u	n	u, y	n	n, u	n	n	n	y	n, n	y	y	u, u	n	n
Aquaculture	Degree of impact	l, m	l	m, l	l	l, m	l	u	l	l	n, l	l	l	l, m	l	n
	Economic loss	u, u	n	Substantial, u	N/A	u, u	u	u	u	not sig	u, n	u	n	n, u	15,000	n
	Legal take	f, f	few	n, few	few	n, few	N/A	u	u	N/A	n, n	few	n	Few, few	u	n
Vegetation	Herbaceous	u, u	n	u, l	y	n, u	p	y	n/u	n	n, n	u	n	n, u	l	n
	Trees	l, y	n	u, l	y	n, n	y	y	n/u	y	n, n	y	l	l, y	u	n
	Rare/unique	n, u	n	u, n	n	n, u	u	n	n/u	n	n, n	u/n	n	n, u	u	n
Other species	Avian	u, n	u	n/u, y	?	?, y	n	n	u	y	u, n	?	n/u	n, n	u	n
	Non-avian	u, n	u	n/u, n	n	n, u	n	n	u	n	n, n	u	n/u	u, n	u	n
Private property		l, n	n	y, n	n	n, y	n	n	n/u	n	n, n	u/n	y	u, n	n	n
Disease transmission		n, u	n	u, n	n	n, u	n	n	n	n	n, n	n	n	n, u	n	n
Water quality		n, ?	n	y, y	n	u, ?	n	n	n/u	n	n, n	n	n	n, ?	n	n
Public health		n, n	n	u, n	n	n, y	n	n	n/u	n	n, n	n	n	n, n	n	n

Summary of fisheries survey data: states / provinces reporting declines linked to DCCOs .

Table 32. Summary of fisheries survey data: states / provinces reporting declines linked to DCCOs.

State/ Province	Declines	Area of Decline	DCCO Link	DCCO data	DCCO Studies	% consumed
DE	yes	Mid-Atlantic	n		n	n
MA	yes	Inshore (marine)	n		n	n
MA	no	Inland/Trout hatcheries	n		n	n
AL	no		n		n	n
IN	no		n		n	n
NJ	variable	Freshwater	n		n	n
ME	yes	Marine	y	anecdotal	y (past)	n
ME	yes	Marine	n		n	n
VT	yes	Freshwater	y	data being analyzed	n	n
MI	yes	Freshwater	y	anecdotal	y (past)	y
AR	unknown				y	n
NY	yes	Freshwater	y	data	y	y
WI	yes	Freshwater	y	% consumed only	y	y
MB	yes	Freshwater	n		y(past)	n
PEI & NB	yes	Freshwater, Marine	n		y	n
GA	yes	Freshwater, Marine	n		n	n
IA	yes	Freshwater	n		n	n
ONT	yes	Freshwater	n	fishermen	y	n
OK	no		n		n	n
CT	yes	Marine	n	On salmon, no impact	n	n
OH	yes	Lake Erie	n		y	n
TN	no		n		n	n
MS	yes	Interior Lakes, MS Delta Region	n		y	small

Table 33. Summary of Fisheries Survey Data: Fish species reported to be in decline and factors thought responsible.

State / Province	Species	Factors Responsible
DE	American shad, black sea bass, scup, bluefish	Overfishing, Loss of critical habitat
MA	scup, tautog, black sea bass, Ocean Pout, Gulf of Maine cod, Cape cod yellowtail, longfinned squid, Gulf of Maine windowpane, Southern New England windowpane	Overexploitation
NJ	Shad and alewife populations are variable	lake trout predation (alewives), Lack of Zooplankton
ME	alewife, blueblack herring, rainbow smelt, winter flounder, Atlantic cod, pollack, sea herring (inshore stocks), Atlantic salmon	Overfishing, Loss of spawning habitat, Increased predation, Pollution, Dams
VT	yellow perch	white perch invasion, Cormorant predation, Zebra mussel
MI	yellow perch, walleye, northern pike, bluegill, smallmouth bass	Overfishing, cormorants, sea lamprey, Natural Cycle, alewife, Zebra mussel, Weather
AR	Unknown if any species are declining	
NY	yellow perch, walleye, smallmouth bass, rock bass	DCCOs, reductions in phosphorus, Zebra mussel
WI	yellow perch, smelt	white perch invasion, cormorants, trout, salmon and burbot predation
MB	walleye, sauger	Overfishing (commercial)
PEI & NB	Eel, Atlantic salmon	Overfishing, Recruitment failure
GA	Savannah River striped bass, Atlantic sturgeon	Overfishing, Habitat degradation
IA	freshwater drum, bullhead	Unknown
ONT	American eel and lake whitefish (L. Ontario); rainbow smelt and white perch (L. Erie); yellow perch (L. Erie, Huron); lake trout (L. Superior)	Global and local environmental change, Temperature, Phosphorus abatement, Zebra mussels, Commercial exploitation
CT	American shad, alewife, blueback herring, winter flounder, tautog	By-catch, Stocked recovery of striped bass, Entrainment, Unknown marine factors
OH	Not provided	Fishing, Exotics, Reduced Productivity
MS	largemouth bass, redear, bluegill	Poor water quality (high silt load in water)

APPENDIX II. CURRENT RESEARCH

To identify current research on DCCOs, we asked recipients of our DCCO and fisheries surveys to identify ongoing DCCO research and population monitoring efforts in their state / province. Recipients were also asked to identify priorities for DCCO research and monitoring. Table 34 shows the states and provinces that reported current research / monitoring, and identified research priorities. Recently completed research was reviewed in earlier sections.

Table 34. States and provinces engaged in current DCCO or DCCO related research or monitoring programs, and or with identified research priorities.

State / Province	Ongoing / Future Research	Currently Monitoring	Research Priority
OK			-Foraging efficiency; -Avg. number/mass of prey consumed daily; -prey selection in small lakes and ponds; -methods to reduce foraging efficiency in ponds and fish hatcheries (M. Howery, OK Dept. Wildl. Cons.)
IA		-Colonial nesting bird survey (E. Kirsch); -Volunteer rookery survey (K. Bogenschutz, IA DNR)	-Continue annual rookery survey (IA DNR)
MI	-Starting study of smallmouth bass population around Beaver Is. due to residents' concerns over DCCO predation (MI DNR Fish. Div. and MI Univ.)	-Monitor nest numbers (G. Belyea, MI DNR)	-Identify and locate commercial fishermen that under report catch (J. Ludwig)
OH	-Telemetry study (1999) to determine where DCCOs are feeding and distances flown on foraging flights on w. Lake Erie (M. Bur, USGS BRD, Sandusky Field Office)	-Population monitoring (M. Shieldcastle, OH Div. of Wildl.)	-Identify problems and solutions to effects on colonial waders -Reduce fisherman concerns (OH Div. of Wildl.)
WI		-Statewide survey every 5 years, and other sites surveyed intermittently (S. Matteson, WI DNR)	

State / Province	Ongoing / Future Research	Currently Monitoring	Research Priority
AL		-Winter surveys for monitoring (F. Boyd, USDA/WS)	-Continue winter surveys -Assist with other research efforts (USDA/WS)
AR	<p>-DCCO impacts on catfish and baitfish production; -Efficacy of harassing DCCOs at individual day roosts and / or night roosts near catfish and baitfish farms; -Determine migratory corridors, breeding locations, and winter movements of DCCOs wintering in catfish production areas of the se U.S. through satellite telemetry study; -Provide information necessary for regional and/or flyway-based DCCO population management strategies; -Quantify and monitor enrichment of nutrients in aquatic and terrestrial environments associated with DCCO night roosts and breeding colonies in AR; -Analyze effects of DCCOs on bottomland hardwood silvics and wetland ecology (S. Werner, WSDA/WS)</p>	-Determine extent of DCCO nesting in se AR (S. Werner, USDA/WS)	-Protect aquaculture -Determine impact on sport fisheries (K. Rowe, AR Game & Fish Comm.; M. Hoy, USDA/WS)
LA		-Monitoring only through CBCs, BBSs, and waterbird colony surveys (B. Vermillion, LA Dept of Wildl. & Fish.)	-Monitor / count birds in Toledo Bend colonies; -Literature search as part of recently developed task-force to investigate cormorant issues; -Review of Migratory Bird Treaty Act. (LA Dept of Wildl. & Fish.)

State / Province	Ongoing / Future Research	Currently Monitoring	Research Priority
MS	<ul style="list-style-type: none"> -Clarifying catfish production losses -Enhancing roost dispersal efforts through shooting (J. Glahn, USDA WS) -Satellite tracking study monitoring movements on wintering grounds and tracking birds back to nesting colonies (S. Werner and Tommy King , USDA / WS) 	<ul style="list-style-type: none"> -Monitoring wintering population and estimating numbers (J. Glahn, USDA WS) 	<ul style="list-style-type: none"> -Determine where, when and how cormorant population control can best be implemented to alleviate DCCO impact on aquaculture along with using cultural and barrier strategies to alleviate impacts -monitor winter numbers and numbers taken under the USFWS depredation order and experimental control procedures(J. Glahn, USDA/WS)
NC		<ul style="list-style-type: none"> -Survey all inland colonies every 5-10 years (D. Allen, NC Wildl. Res. Comm.) 	
SC		<ul style="list-style-type: none"> -Periodic ground counts of nests; -Annual aerial estimates of nest numbers (T. Murphy, SC DNR) 	
ME		<ul style="list-style-type: none"> -Monitor colonial seabirds nesting on coastal islands (R.B. Allen, ME Dept. Inland Fish & Wildl.) 	<ul style="list-style-type: none"> -Continue monitoring (R.B. Allen)
MD		<ul style="list-style-type: none"> -Periodic population census every 3-4 years (D. Brinker, MD DNR) 	<ul style="list-style-type: none"> -Population monitoring (D. Brinker)
MA		<ul style="list-style-type: none"> -Statewide nesting survey at 10 year intervals, next in 2004-2005 (B. Blodget, MA Div. of Fish & Wildl.) 	<ul style="list-style-type: none"> -Continue monitoring breeding colonies every 10 years to document trends (B. Blodget) -Measure damage from droppings to habitat; -Measure damage from feeding on herring runs; -Measure damage to water quality; -Measure damage to property values; -Measure damage to human

State / Province	Ongoing / Future Research	Currently Monitoring	Research Priority
			health; -Study habitat use preferences (roosts, nesting areas) -Study roost dispersal effectiveness techniques (L. Henze, USDA / WS)
NY	-Assessing impacts to Oneida Lake yellow perch and walleye populations through population monitoring (fisheries and joint effort with DCCO population / diet monitoring) and fisheries assessment of current DCCO management (reduction in successful management and harassment of fall migrants) (L. Rudstam, Cornell Biological Field Station 315-633-2358) -Assessing impacts to Lake Ontario smallmouth bass population through population monitoring and DCCO diet assessment (joint effort with wildlife staff on DCCO population monitoring) and effectiveness of egg oiling (A. Schiavone, NYSDEC-Fisheries 315 785-2621). -Determine foraging range and behavior of DCCOs nesting on Oneida Lake (J. Coleman, NY Coop Fish & Wildl. Res. Unit, Cornell Univ)	-Monitor upstate nesting colonies annually, and Long Island colonies every 3 years (R. Miller, NYSDEC)	-Continue monitoring nesting colonies -Conduct research to determine effects of DCCO control efforts at Lake Ontario and Oneida Lake islands (R. Miller)
PA		-Nesting colony is monitored (D. Brauning, PA Game Comm.)	-Monitor nesting population (D. Brauning, PA Game Comm.)

State / Province	Ongoing / Future Research	Currently Monitoring	Research Priority
RI	-Studies to assess impacts to commercial and sport fisheries (M. Gibson, Wickford Marine Base)	-Annual nest counts (C. Raithel, RI Div Fish & Wildl.)	-Assess potential impact to fisheries (C. Raithel, RI Div Fish & Wildl.)
VT		-Monitor islands for nesting, and annual nest counts at Lake Champlain (L. Garland, VT Dept Fish & Wildl.)	
MB		-Monitor breeding population via nest counts (B. Koonz, MB DNR)	
NB	Study on predation of Atlantic salmon smolts by mergansers, cormorants and seals in New Brunswick (F. Whoriskey, Atlantic salmon Federation, emerrill@nbnet.nb.ca)		
ONT	-Diet study underway this summer at e. Lake Ontario colonies (Alastair Mathers, Lake Ont Mgmt Unit, Ont. Min. Nat. Res. 613-476-8733)	-Monitor numbers on Great Lakes, also some periodic inland counts (J. Harcus, Ont. Min. Nat. Res.)	-Cormorant food studies in e. Lake Ontario for 1999 if funds are available; -Studies to assess impacts to fish stocks in inland lakes; -More standardized monitoring of DCCO numbers across the province in areas other than the Great Lakes (J. Harcus, Ont. Min. Nat. Res.)
PEI	-Bioenergetic modeling for consumption estimates of Atlantic salmon by avian predators (cormorants, mergansers and kingfishers) in the maritime provinces, as a percent of standing stock (D. Cairns, Dept of Fisheries & Oceans, PEI);		

APPENDIX III. LIST OF COMMON AND SCIENTIFIC NAMES

BIRDS

Common Name	Scientific Name	Abbreviation
Anhinga	<i>Anhinga anhinga</i>	
American Avocet	<i>Recurvirostra americana</i>	
American Black Duck	<i>Anas rubripes</i>	
American White Pelican	<i>Pelecanus erythrorhynchos</i>	
Bald Eagle	<i>Haliaeetus leucocephalus</i>	
Canada Goose	<i>Branta canadensis</i>	
Common Eider	<i>Somateria mollissima</i>	
Common Raven	<i>Corvus corax</i>	
cormorants:		
Brandt's Cormorant	<i>Phalacrocorax penicillatus</i>	
Double-crested Cormorant	<i>P. auritus</i>	DCCO
Great Cormorant	<i>P. carbo</i>	GRCO
Neotropic Cormorant	<i>P. brasilianus</i>	
Pelagic Cormorant	<i>P. pelagicus</i>	
Red-faced Cormorant	<i>P. urile</i>	
crow sp.	<i>Corvus sp.</i>	
egrets:		
Cattle Egret	<i>Bubulcus ibis</i>	
Great Egret	<i>Ardea alba</i>	
Gadwall	<i>Anas strepera</i>	
gulls:		
California Gull	<i>Larus californicus</i>	
Greater Black-backed Gull	<i>L. marinus</i>	
Herring Gull	<i>L. argentatus</i>	
Ring-billed Gull	<i>L. delewarensis</i>	
Gyr Falcon	<i>Falco rusticolus</i>	
herons:		
Black-crowned Night-heron	<i>Nycticorax nycticorax</i>	BCNH
Great Blue Heron	<i>Ardea herodias</i>	GRBH
Little Blue Heron	<i>Egretta caerulea</i>	
Mallard	<i>Anas platyrhynchos</i>	
Osprey	<i>Pandion haliaetus</i>	
Oystercatcher	<i>Haematopus ostralegus</i>	
plover sp.	<i>Pluvialis sp.</i> and <i>Charadrius sp.</i>	
terns:		
Caspian Tern	<i>Sterna caspia</i>	
Common Tern	<i>Sterna hirundo</i>	
swallow sp.	<i>Hirundinidae</i>	

FISHES

Common names of fishes are not traditionally capitalized in publications (American Fisheries Society 1991), whereas standard format for bird common names (American Ornithologists' Union 1998) requires use of capital letters. Use of these two different formats respects nomenclature decisions followed by these two professional societies. Species have been grouped together when common names reflect taxonomic relationships; otherwise, they are alphabetized by first letter of the common name.

Common Name	Scientific Name
alewife	<i>Alosa pseudoharengus</i>
Atlantic menhaden	<i>Brevoortia tyrannus</i>
bass sp.	PERCIFORMES
largemouth bass	<i>Micropterus salmoides</i>
rock bass	<i>Ambloplites rupestris</i>
smallmouth bass	<i>Micropterus dolomieu</i>
striped bass	<i>Morone saxatilis</i>
bloater	<i>Coregonus hoyi</i>
bluegill	<i>Lepomis macrochirus</i>
bluefish	<i>Pomatomus saltatrix</i>
bream	bluegill or pinfish
bullhead sp.	Ictaluridae (<i>Ameiurus sp.</i>)
burbot	<i>Lota lota</i>
capelin	<i>Mallotus villosus</i>
carp sp.	Cyprinidae
common carp	<i>Cyprinus carpio</i>
catfish, sp.	Ictaluridae
channel catfish	<i>Ictalurus punctatus</i>
chub sp.	Cyprinidae
lake (northern) chub	<i>Couesius plumbeus</i>
cisco (also lake herring)	<i>Coregonus artedi</i>
cod sp.	Gadidae
Atlantic cod	<i>Gadus morhua</i>
crappie sp.	Centrarchidae (<i>Pomoxis sp.</i>)
black crappie	<i>Pomoxis nigromaculatus</i>
white crappie	<i>Pomoxis annularis</i>
cunner	<i>Tautoglabrus adspersus</i>
drum sp.	Sciaenidae
freshwater drum	<i>Aplodinotus grunniens</i>
eel sp.	ANGUILLIFORMES
American eel	<i>Anguilla rostrata</i>
flounder sp.	Bothidae and Pleuronectidae
winter flounder	<i>Pleuronectes americanus</i>

herring sp.
 Atlantic herring
 blueback herring
 lake herring
 Pacific herring
 sea herring
 gunnel sp.
 crescent gunnel
 penpoint gunnel
 lake herring (also cisco)
 lamprey sp.
 sea lamprey
 mackerel
 northern pike
 peamouth
 pinfish
 pollock
 pout
 ocean pout
 prickleback sp.
 snake prickleback
 pumpkinseed
 redeer
 salmon, sp.
 Atlantic salmon
 sand lance sp.
 American sand lance
 Pacific sand lance
 sauger
 sculpin sp.
 slimy sculpin
 spoonhead sculpin
 sea bass sp.
 black sea bass
 seapearch sp.
 striped seapearch
 scup
 shad sp.
 American shad
 gizzard shad
 threadfin shad
 shiner perch
 shiner sp.
 common shiner
 emerald shiner
 spottail shiner

Clupeidae
 Clupea harengus
 Alosa aestivalis
 see “lake herring (also cisco)”
 Clupea pallasii
 see Atlantic herring
 Pholidae
 Pholis laeta
 Apodichthys flavidus
 Salmonidae, *Coregonus artedii*
 Petromyzontidae
 Petromyzon marinus
 Scombridae
 Esox lucius
 Mylocheilus caurinus
 Lagodon rhomboides
 Pollachius virens
 sp. Zoarcidae
 Macrozoarces americanus
 Stichaeidae
 lumpenus sagitta
 Lepomis gibbosus
 see redeer sunfish
 Salmonidae
 Salmo salar
 Ammodytidae
 Ammodytes americanus
 Ammodytes hexapterus
 Stizostedion canadense
 Cottidae
 Cottus cognatus
 Cottus ricei
 Serranidae
 Centropristis striata
 Embiotocidae
 Embiotoca lateralis
 Stenotomus chrysops
 Clupeidae
 Alosa sapidissima
 Dorosoma cepedianum
 Dorosoma petenense
 Cymatogaster aggregata
 Cyprinidae
 Luxilus cornutus
 Notropis atherinoides
 Notropis hudsonius

smelt sp.	Osmeridae
rainbow smelt	<i>Osmerus mordax</i>
splake	hybrid trout
	(<i>Salvelinus fontinalis</i> x <i>S. namaycush</i>)
stickleback sp.	Gasterosteidae
ninespine stickleback	<i>Pungitius pungitius</i>
threespine stickleback	<i>Gasterosteus aculeatus</i>
sturgeon sp.	Acipenseridae
Atlantic sturgeon	<i>Acipenser oxyrhynchus</i>
sucker sp.	Catostomidae
white sucker	<i>Catostomus commersoni</i>
sunfish sp.	Centrarchidae
green sunfish	<i>Lepomis cyanellus</i>
redeer sunfish	<i>Lepomis microlophus</i>
tautog	<i>Tautoga onitis</i>
tilapia sp.	Cichlidae (<i>Tilapia</i> sp.)
blue tilapia	<i>Tilapia aurea</i>
tomcod sp.	Microgadus sp.
Atlantic tomcod	<i>Microgadus tomcod</i>
trout sp.	Salmonidae
brook trout	<i>Salvelinus fontinalis</i>
lake trout	<i>Salvelinus namaycush</i>
trout-perch	<i>Percopsis omiscomaycus</i>
	or <i>P. transmontana</i>
yellow perch	<i>Perca flavescens</i>
walleye	<i>Stizostedion vitreum</i>
whitefish	Salmonidae
lake whitefish	<i>Coregonus clupeaformis</i>
round whitefish	<i>Prosopium cylindraceum</i>
white perch	<i>Morone americana</i>
windowpane	<i>Scophthalmus aquosus</i>
yellowtail	<i>Seriola lalandi</i>

MAMMALS

Common Name	Scientific Name
arctic fox	<i>Alopex lagopus</i>
ground squirrel	<i>Spermophilus undulatus</i>
Norway rat	<i>Rattus norvegicus</i>
rabbit	<i>Oryctolagus cuniculus</i>
red fox	<i>Vulpes vulpes</i>

ARTHROPODS

Common Name	Scientific Name
sand shrimp (also common shrimp)	<i>Crangon crangon</i>
shrimp sp.	Crustacea
crayfish / crawfish	<i>Procambarus clarkii</i>
crayfish / yabbie	<i>Cherax destructor</i>

OTHER INVERTEBRATES

Common Name	Scientific Name
Dreissenid mussels	<i>Dreissena sp.</i>
Longfinned squid	<i>Loligo pealei</i>
Zebra mussel	<i>Dreissena polymorpha</i>

PLANTS

Common Name	Scientific Name
cottonwood sp.	<i>Populus sp.</i>
Canadian milk vetch	<i>Astragalus canadensis</i>
sandbar willow	<i>salix exigua</i>
stinging nettle	<i>Urtica dioica</i>
thistle sp.	<i>Cirsium sp.</i>

APPENDIX IV. LIST OF ABBREVIATIONS

ADC	Animal Damage Control
APHIS	Animal and Plant Health Inspection Service
BMR	basal metabolic rate
CBC	(National Audubon Society) Christmas Bird Count
CPUE	catch per unit effort
DFI	daily food intake
DNR	Department of Natural Resources
EIS	environmental impact assessment
kJ	kilojoule (measure of energy)
MBTA	Migratory Bird Treaty Act
NASS	National Agricultural Statistics Service
NGO	non-governmental organization
NPS	National Park Service
NWR	national wildlife refuge
NYSDEC	New York State Department of Environmental Conservation
PRBO	Point Reyes Bird Observatory
USACE	United States Army Corps of Engineers
USDA/WS	United States Department of Agriculture / Wildlife Services
USGS	United States Geographical Survey
USFWS	United States Fish and Wildlife Service
VDFW	Vermont Department of Fisheries and Wildlife
WMA	wildlife management area

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