For monitoring purposes, habitat type has been all important (Messel *et al.* 1981; Webb *et al.* 1986). Most spotlight surveys have been carried out in tidal rivers, where boat access is possible and where at low tide, crocodiles tend to be at the edge of the mudbanks below fringing vegetation - their eyes are not shielded from the spotlight by vegetation. In heavily vegetated, non-tidal freshwater swamps, spotlight surveying is of limited value because unless a crocodile has positioned itself in a patch of open water, the vegetation shields the eyes from the light. Heavily vegetated freshwater swamps contain significant saltwater crocodile populations and are some of the main habitats used by saltwater crocodiles for nesting (Webb *et al.* 1983f). During the period of extensive hunting these habitats provided significant refuges for crocodiles, because it was extremely difficult for hunters to get access to the crocodiles.

## 3. THE SITUATION FROM 1971-77

Although a variety of "spotcheck" day and night surveys were carried out in various rivers between 1971 and 1974, it was not until 1974-75 that standardised spotlight surveys were developed and a more systematic survey program was introduced. There are no survey data from 1971. However, an intensive mark-recapture program undertaken between 1973 and 1976 (Fig. 3), and size estimates made during spotlight counts in 1974-75, allow general identification of animals born before or after 1971. The general situation was that adults were very rare and extremely wary (Webb and Messel 1979), and only in remote areas were there any juvenile size classes left; they were collected each year in river systems that were easily accessible. The lack of adults and the alteration of nesting sites due to overgrazing by buffalo and cattle, and through saltwater intrusion following overgrazing, had greatly restricted nesting sites in some areas (Hill and Webb 1982; Letts *et al.* 1979).

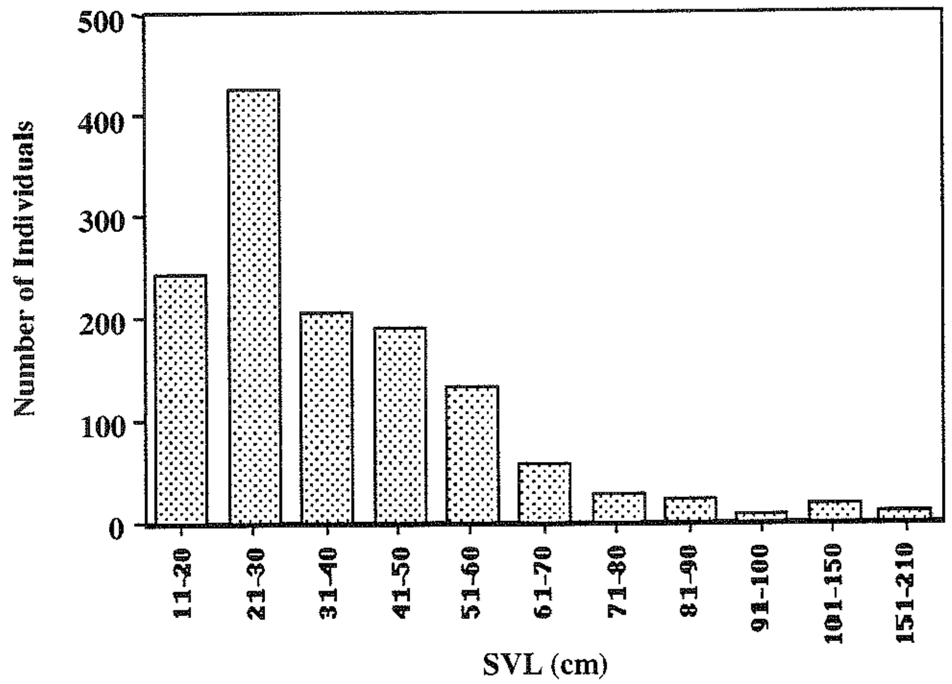


Figure 3. Size distribution (SVL; snout-vent length) for 1354 *C. porosus* caught in the Northern Territory between 1973 and 1976 (after Webb and Messel 1978a). Most crocodiles less than 150 cm hatched around the time of protection or since protection.



In the immediate post-protection years, despite high egg mortality generally (around 75%; Webb and Manolis 1989), hatchlings started to appear in tidal rivers. These came from riverside nests (Webb *et al.* 1977) and from freshwater swamps (Magnusson 1979a). The immediate post-protection hatchlings had high survival rates, probably due in part to the low densities of larger crocodiles. After the first few years of expanding juvenile numbers, the survival rates for these younger size classes started to decline (Messel *et al.* 1981a, 1984, 1986d, 1987) due to density-dependent mechanisms such as cannibalism and social exclusion (Webb and Manolis 1991). With social exclusion, stimulating emigration from nesting areas, came the spread of young crocodiles into areas where no nesting occurred. That is, some rivers started to be repopulated through local breeding whereas others depended on immigration. Size estimates made during widespread spotlight counts across the NT in 1975-76 confirmed the general trends, with some 80% of all crocodiles sighted (N= 1437) having been hatched at or since the time of protection.

Areas close to Darwin (eg Adelaide R., Mary R.) had traditionally seen the highest hunting pressures. Adults and juveniles were at very low densities and nesting habitats were badly affected by overgrazing. The initial recovery was slow and depended on immigration (Webb and Messel 1978b). Other areas such as the Finniss and Reynolds River systems, with large amounts of freshwater swamp, and the Liverpool and Tomkinson Rivers in remote parts of Arnhem Land, had greater numbers of remaining adults, some juvenile cohorts at protection, and intact nesting habitat: the population started to increase mainly through recruitment from local breeding.

In any overview, the first few years of protection saw a dramatic increase in the number of individuals in the wild population (400% by 1975-76), but they were mainly hatchlings and 1 to 4-year olds (Fig. 3). Adult numbers remained low but stable. Given that 12-16 years would be required before wild *C. porosus* reached maturity (Webb *et al.* 1978a), it would not be until the mid-1980s that *C. porosus* hatched since protection would finally start to breed themselves.

## 4. CHANGING SURVIVAL RATES PROMOTED RAPID RECOVERY

That the rate of expansion of the recovering saltwater crocodile population would decline dramatically over time was a function of changing survival and movement rates. Messel *et al.* (1981a, 1984, 1986d, 1987) were the first to identify that density-dependent mechanisms were vitally important. They hypothesised that there were a set number of spaces in the river that could be occupied by crocodiles, and that once these were filled, the rate of population increase was greatly constrained. When analysing the spotlight count data for the Blyth-Cadell River system between 1974 and 1990, Webb and Manolis (1991) were able to demonstrate that in the first three years of life, three different density-dependent mechanisms promoting fast recovery from a depleted population status were operating.

Between hatching and one year of age, survival rates were negatively correlated with the number of hatchlings in the river. If there were few hatchlings entering a river they had high survival rates to one year of age (100 hatchlings; 80% survival), but if there were many hatchlings, the survival rate decreased dramatically (450 hatchlings; 20% survival). However, the rates themselves were independent of the numbers of other crocodiles present. That is, it probably reflects predation by non-crocodilian predators (birds, fish, snakes, varanid lizards, etc.).

In contrast, the survival rate between 1 and 2 years of age were highly correlated with the numbers of large (greater than 2 m) crocodiles in the river, which is consistent with known cannibalism. Between 2-3 and 3-4 years of age, the retention of crocodiles in a river was highly negatively correlated with the total numbers of non-hatchlings in the river, suggesting "space" was limiting. All these density-dependent factors favoured a rapid increase in the population when it was severely depleted, with progressive constraints on expansion rate as the population started to build. That the crocodiles which moved out of many of these rivers were numerically far greater than the numbers which built up in other rivers led Messel *et al.* (1981) to conclude that movement out of the rivers was itself associated with a much higher mortality rate than was being experienced within the rivers.

## 5. LATER YEARS OF RECOVERY (1977-98): SPOTLIGHT COUNTS

### 5.1. General

Two major population census methods were employed in the Northern Territory: spotlight counts in tidal rivers, following the methods described by Messel *et al.* (1981a); and, helicopter counts over sample segments of river, following methods described by Bayliss *et al.* (1986). In 1989, when the monitoring programs were rationalised, two separate survey programs were undertaken for two totally different reasons, but some confusion has developed over this.

Spotlight counts in tidal rivers involve the complete navigable length of the river (often exceeding 100 km), and are precise and accurate. But they are costly and time consuming to conduct. In the NT they were restricted to a small number of high density rivers and were designed to provide detailed information on the numbers and size structure of the population in those rivers, so that the recovery of the population in high density rivers could be examined in detail.

Given the heterogeneous nature of different rivers with regard to density and carrying capacity, extrapolating trends from these sample rivers to the complete NT population proved problematic (Webb *et al.* 1984, 1989). Hence, helicopter surveys were used to monitor small sample segments (10 km) in a much wider range of low, medium and high density rivers (N= 70), providing a direct and cost-effective index of whether the total population was increasing, decreasing or stable.

The two methods were never designed to be comparable in terms of the accuracy or precision with which the status of crocodiles in any one river is revealed by a survey (eg spotlight count over 100 km versus a helicopter count along one side of the river for 10 km), and any comparisons are invalid. Helicopter counts are a precise and cost-effective survey method (Bayliss *et al.* 1987; Webb *et al.* 1986, 1989), and one which allows a much higher proportion of the large crocodiles (often seen as "eyes only" in spotlight surveys) to be sighted.

### 5.2. Methods

To examine general trends in abundance and size structure of the saltwater crocodile population as revealed by spotlight counts, a matrix of results from 11 major river systems was created spanning a period of 22 years (Table 1). Most of these rivers were surveyed in most years, but where gaps existed, they were replaced with predicted values derived from regression analysis. The use of predicted values adds error to specific specific rivers, but reduces much more significant biases associated with heterogeneous densities in different rivers.



Table 1. Summary of spotlight data for major NT river survey units. Absolute numbers (No.) and densities (Dens.) are provided for the years 1977-8 and 1984-98. Numbers in parentheses are predicted values (see text).

| | 1977 | | 1978 | | 1984 | | 1985 | | 1986 | | 1987 | | 1988 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* |
| **Adelaide 2** | 113 | 1.03 | 88 | 0.85 | 147 | 1.21 | 80 | 0.70 | 92 | 0.81 | 144 | 1.41 | 132 | 1.19 |
| **Adelaide 3** | 90 | 1.71 | 81 | 1.54 | 158 | 3.01 | 171 | 3.26 | 180 | 3.43 | 177 | 3.37 | 239 | 4.55 |
| **Adelaide 4** | 184 | 2.85 | 189 | 2.93 | 233 | 3.61 | 194 | 3.01 | 201 | 3.12 | 205 | 3.18 | 324 | 5.02 |
| **Blyth 1** | 168 | 3.37 | 145 | 2.91 | (185) | (3.71) | 125 | 2.51 | 173 | 3.47 | 249 | 5.00 | (200) | (4.02) |
| **Blyth 2** | 17 | 1.36 | 18 | 1.44 | (13) | (1.13) | (13) | (1.13) | 6 | 0.83 | 16 | 1.52 | (12) | (1.04) |
| **Cadell 1** | 79.5 | 2.76 | 81 | 2.87 | (86.4) | (2.85) | 55 | 1.88 | 93 | 3.10 | 94 | 3.16 | 81 | 2.73 |
| **Daly 1** | (85) | (0.94) | 116 | 1.22 | 180 | 1.93 | 221 | 2.35 | 265 | 2.81 | 289 | 3.07 | 270 | 2.87 |
| **East Alligator 1** | 124 | 2.64 | 152 | 2.99 | 205 | 4.18 | 217 | 5.23 | (218) | (4.64) | 249 | 6.04 | 307 | 7.45 |
| **Liverpool 1** | 79 | 1.32 | 90 | 1.58 | 117 | 1.77 | (121) | (2.02) | 119 | 1.98 | 128 | 2.29 | 146 | 2.56 |
| **Mary 2** | (5) | (0.30) | (6) | (0.36) | 26 | 1.53 | 25 | 1.50 | 40 | 2.40 | 36 | 2.16 | 65 | 3.89 |
| **Mary 3** | (18) | (0.80) | (23) | (1.02) | 94 | 3.64 | 89 | 3.99 | 104 | 4.66 | 143 | 6.41 | 134 | 6.01 |
| **South Alligator** | 87 | 1.34 | 79 | 1.22 | 162 | 2.49 | (158) | (2.6) | (167) | (2.75) | 181 | 2.78 | 202 | 3.37 |
| **Tomkinson** | 58 | 1.09 | 81 | 1.53 | (109) | (2.04) | (112) | (2.11) | 104 | 1.83 | 126 | 2.38 | (119) | (2.24) |
| **West Alligator** | 57 | 1.50 | 53 | 1.39 | 96 | 2.59 | (95) | (2.59) | (98) | (2.69) | 88 | 2.50 | 107 | 2.97 |
| **Wildman** | (105) | (3.31) | 65 | 1.94 | 200 | 5.97 | (149) | (4.65) | 123 | 3.78 | 157 | 4.69 | 196 | 6.13 |
| **N** | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 |
| **Mean** | 85 | 1.76 | 84 | 1.72 | 134 | 2.78 | 122 | 2.64 | 132 | 2.82 | 152 | 3.33 | 169 | 3.74 |
| **StDev** | 51.1 | 0.97 | 50.7 | 0.83 | 64.3 | 1.30 | 64.3 | 1.25 | 68.4 | 1.15 | 76.9 | 1.53 | 90.1 | 1.83 |
| **StErr** | 13.2 | 0.25 | 13.1 | 0.22 | 16.6 | 0.34 | 16.6 | 0.32 | 17.7 | 0.30 | 19.8 | 0.40 | 23.3 | 0.47 |
| **Min** | 5 | 0.30 | 6 | 0.36 | 13 | 1.13 | 13 | 0.70 | 6 | 0.81 | 16 | 1.41 | 12 | 1.04 |
| **Max** | 184 | 3.37 | 189 | 2.99 | 233 | 5.97 | 221 | 5.23 | 265 | 4.66 | 289 | 6.41 | 324 | 7.45 |

| | 1992 | | 1993 | | 1994 | | 1995 | | 1996 | | 1997 | | 1998 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* | *No.* | *Dens.* |
| **Adelaide 2** | 126 | 1.12 | 106 | 0.94 | 115 | 1.23 | 109 | 1.16 | 109 | 1.01 | 173 | 1.60 | (134) | (1.22) |
| **Adelaide 3** | 291 | 5.54 | 357 | 6.80 | 273 | 5.20 | 235 | 4.48 | 246 | 4.69 | 243 | 4.63 | (278) | (5.30) |
| **Adelaide 4** | 262 | 4.06 | 317 | 4.91 | 212 | 3.29 | 287 | 4.45 | 244 | 3.78 | 309 | 4.79 | (273) | (4.24) |
| **Blyth 1** | 191 | 3.83 | 215 | 4.31 | 247 | 4.95 | 209 | 4.19 | 175 | 3.51 | 263 | 5.27 | 225 | 4.51 |
| **Blyth 2** | 13 | 1.49 | 7 | 0.70 | 8 | 0.55 | 10 | 0.69 | 9 | 0.96 | 15 | 1.26 | 15 | 1.03 |
| **Cadell 1** | 109 | 4.72 | 76 | 2.56 | 73 | 2.46 | 95 | 3.20 | 88 | 2.96 | 51 | 1.89 | (81.8) | (3.31) |
| **Daly 1** | 386 | 4.28 | 388 | 4.30 | 345 | 3.82 | 353 | 4.14 | 340 | 3.99 | 451 | 5.29 | 391 | 4.59 |
| **East Alligator 1** | (247) | (5.26) | (249) | (5.3) | (250) | (5.32) | 231 | 4.36 | 250 | 5.10 | 227 | 4.88 | 252 | 4.99 |
| **Liverpool 1** | 177 | 2.95 | 149 | 2.48 | 187 | 3.12 | 128 | 2.13 | 124 | 2.07 | 167 | 2.78 | (158.7) | (2.65) |
| **Mary 2** | 53 | 3.17 | 55 | 3.29 | 71 | 4.25 | 82 | 4.91 | 99 | 5.93 | 98 | 5.87 | 132 | 7.90 |
| **Mary 3** | 249 | 11.17 | 379 | 17 | 397 | 17.8 | 396 | 17.76 | 312 | 13.99 | 444 | 19.91 | 354 | 15.87 |
| **South Alligator** | (222) | (3.65) | (231) | (3.8) | (241) | (3.96) | 243 | 4.26 | 277 | 4.95 | 297 | 5.21 | 226 | 3.96 |
| **Tomkinson** | 110 | 2.27 | 114 | 2.24 | 140 | 2.56 | 124 | 2.27 | 82 | 1.56 | 134 | 2.41 | (107) | (2.01) |
| **West Alligator** | (113) | (3.08) | (114) | (3.12) | (115) | (3.15) | 120 | 3.33 | 122 | 3.39 | 115 | 3.19 | 111 | 3.08 |
| **Wildman** | (186) | (5.82) | (191) | (5.98) | (197) | (6.15) | 163 | 5.82 | 203 | 6.34 | 223 | 7.19 | 226 | 7.06 |
| **N** | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 | 15 |
| **Mean** | 182 | 4.16 | 197 | 4.52 | 191 | 4.52 | 186 | 4.48 | 179 | 4.28 | 214 | 5.08 | 198 | 4.78 |
| **StDev** | 97.8 | 2.38 | 122.2 | 3.86 | 106.4 | 3.98 | 106.4 | 3.94 | 97.0 | 3.16 | 127.7 | 4.46 | 104.1 | 3.60 |
| **StErr** | 25.3 | 0.61 | 31.6 | 1.00 | 27.5 | 1.03 | 27.5 | 1.02 | 25.0 | 0.82 | 33.0 | 1.15 | 27.9 | 0.93 |
| **Min** | 13 | 1.12 | 7 | 0.70 | 8 | 0.55 | 10 | 0.69 | 9 | 0.96 | 15 | 1.26 | 15 | 1.03 |
| **Max** | 386 | 11.17 | 388 | 17.00 | 397 | 17.80 | 396 | 17.76 | 340 | 13.99 | 451 | 19.91 | 391 | 15.87 |

Table 2. Summary of spotlight surveys from 11 major river systems in the NT. NH tot = total non-hatchlings in all surveys; Dns tot = overall mean density; 2-4', 4-6', >6' = total crocodiles in size class in all surveys; EO = total eyeshines;

| Year | km | NH tot | Dns tot | 2-4' | 4-6' | >6' | EO |
|---|---|---|---|---|---|---|---|
| 1977 | 746.3 | 1270 | 1.70 | 427 | 446 | 241 | 157 |
| 1978 | 743.2 | 1267 | 1.70 | 394 | 427 | 276 | 171 |
| 1984 | 770.1 | 2011 | 2.61 | 624 | 390 | 452 | 545 |
| 1985 | 738.6 | 1825 | 2.47 | 378 | 352 | 482 | 613 |
| 1986 | 744.7 | 1983 | 2.66 | 440 | 363 | 599 | 583 |
| 1987 | 726.5 | 2282 | 3.14 | 530 | 321 | 742 | 689 |
| 1988 | 731.6 | 2534 | 3.46 | 670 | 364 | 733 | 769 |
| 1992 | 725.5 | 2735 | 3.77 | 589 | 355 | 825 | 968 |
| 1993 | 735.9 | 2949 | 4.01 | 584 | 358 | 900 | 1108 |
| 1994 | 725 | 2871 | 3.96 | 573 | 492 | 872 | 933 |
| 1995 | 717.7 | 2785 | 3.88 | 508 | 391 | 915 | 971 |
| 1996 | 724 | 2680 | 3.70 | 376 | 397 | 956 | 951 |
| 1997 | 724.2 | 3210 | 4.43 | 491 | 490 | 1153 | 1076 |
| 1998 | 728.5 | 2965 | 4.07 | 476 | 395 | 1210 | 981 |

## 5.3. Densities

The densities (Fig. 4) recorded from the rivers surveyed (Table 1) indicate that one survey section in the Mary River (Sampan Creek-Shady Camp), had densities greater than two standard deviations from the means of the other rivers surveyed. The pattern of recovery in this river was clearly very different and it has been excluded from the general trend analyses and is discussed separately.

For the 11 major Northern Territory rivers (Table 1), the relationship between density and time immediately following protection (1971-77) is not known precisely, because few of the rivers were surveyed during this period. Given that a 400% increase in density occurred in these first years (see above), the mean density at the time of protection (1971) was around 0.3-0.4 non-hatchlings per kilometre, and the pattern of increase is assumed to have followed that described by Webb *et al.* (1984).

The relationship between density and years since protection (YSP) between 1977 and 1998 shows a significant ($r^2$= 0.96, p= 0.0001) and sigmoidal pattern of increase. Three stages can be identified: relatively low densities in 1977 and 1978 (6 and 7 YSP); a high rate of density increase up to around 20 YSP (1991-92); and, a low rate of increase between 20-21 YSP and 26-27 YSP (1997-98). The polynomial regression is an approximation of the trend, which could be equally modelled with two straight lines. These results suggest that the mean non-hatchling density of *C. porosus* in the major tidal river systems in the Northern Territory is approaching or has reached a plateau.

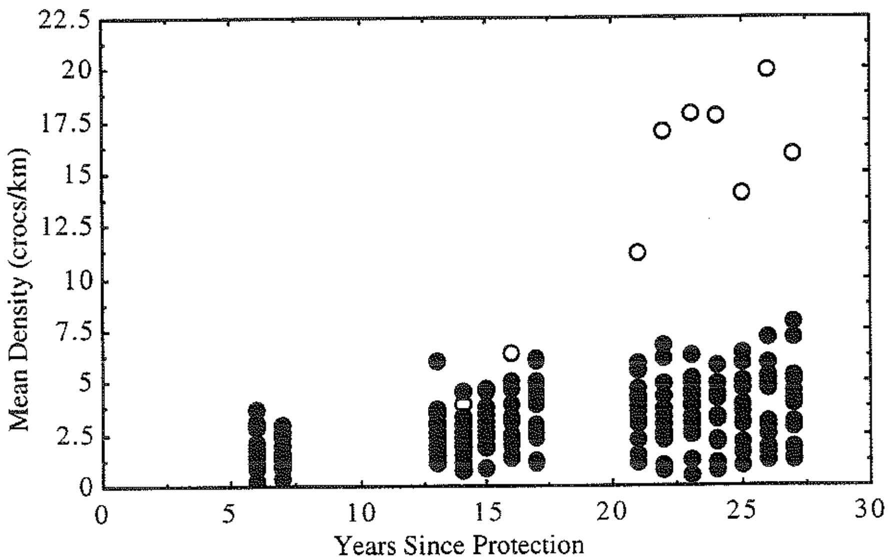


Figure 4. Mean density of non-hatchling *C. porosus* sighted during spotlight counts in 15 survey areas in 11 major rivers in the Northern Territory between 1977 (6 years after protection) and 1998 (27 years after protection). Raw data are in Table 1. Densities recorded (and predicted) from Sampan Creek-Shady Camp section of the Mary River are represented by open circles. Years Since Protection: 1= 1972; 27= 1998.



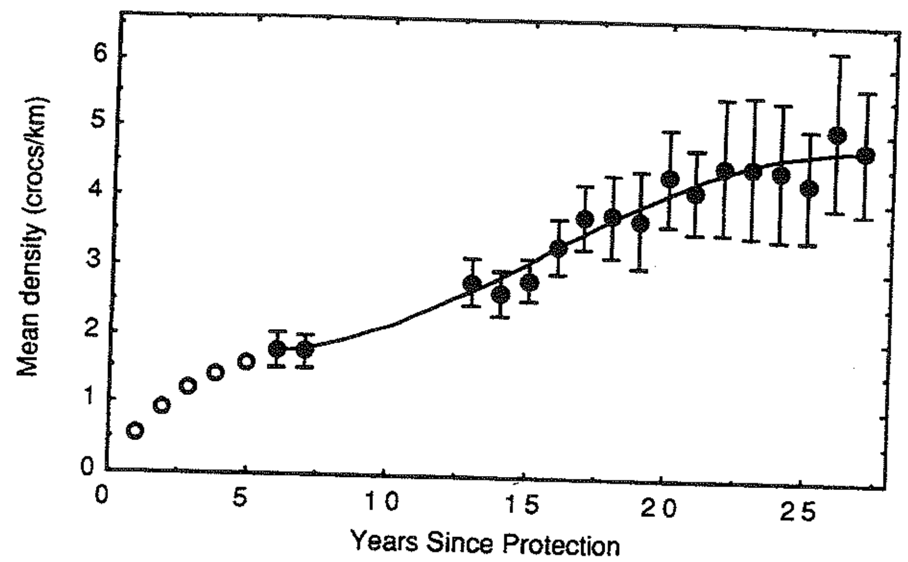


Figure 5. Mean (mean of means) non-hatchling density (crocs/km) for the combined spotlight count data (Table 1) from the Daly, Adelaide, Mary (excluding Sampan Creek-Shady Camp section), Wildman, West Alligator, South Alligator, East Alligator, Liverpool, Tomkinson, Blyth and Cadell Rivers grouped (N= 14 survey areas in 11 rivers). Vertical bars represent one standard error from the mean. The line is the third order polynomial regression of best fit ($r^2$= 0.917, p= 0.0001). Years Since Protection: 1= 1972; 27= 1998. Open dots reflect the general trends thought to have occurred between 1972 and 1976.

## 5.4. Size Structure

Within spotlight counts, crocodiles recorded as "eyes only" represent a significant bias. Calibrations between spotlight and helicopter counts confirm that most "eyes only" are large animals (Webb *et al.* 1989), and the strong correlation (Fig. 6) between the number of >6' animals sighted (Table 2) and the numbers of "eyes only" is generally consistent with this, although results suggest wariness in larger animals is gradually declining. For these analyses all "eyes only" data were assigned to the >6' size class (ie >6'+EO).

The size structure of *C. porosus* sighted during spotlight surveys in the 11 major tidal river systems has continued to change over time (Fig. 7). The >6'+EO size class has shown a high and continuing mean rate of density increase between 1977 and 1998 ($r^2$= 0.97, p= 0.0001: linear regression), which equates to about 0.15 crocodiles/km/year (regression slope).

In contrast, after having increased dramatically in the immediate post-protection period (1971-77), the smaller size classes (2-4' and 4-6') have since shown a high degree of stability over the same time. The 4-6' size class remained stable between 1977 and 1990, but then increased significantly between 1991 and 1998 ($r^2$= 0.56, p= 0.0031, second order polynomial regression). The density of 2-4' crocodiles increased between 1977 and 1991, but then declined, although the trend was not statistically significant. The trends are equally pronounced when expressed as percentage composition of the sighted population (Fig. 8).

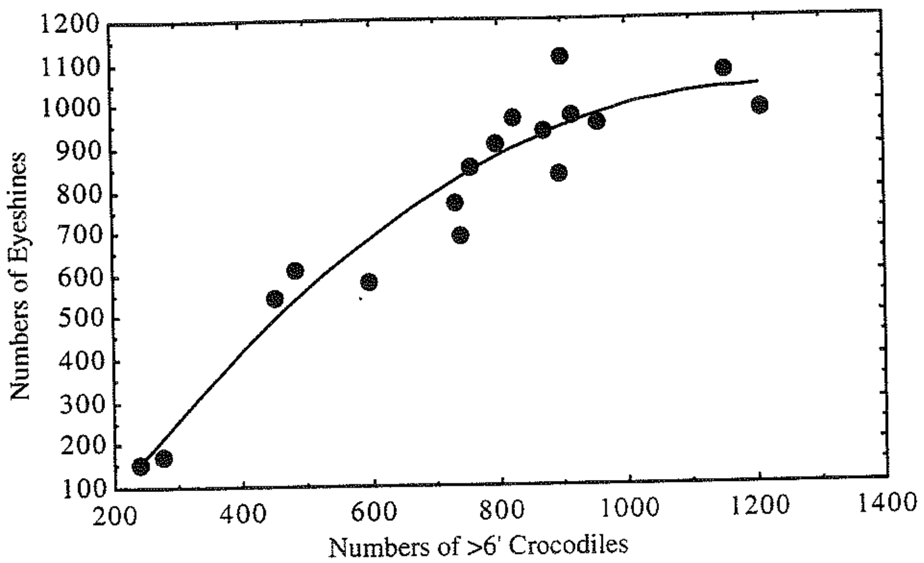


Figure 6. Relationship between the number of >6' *C. porosus* and "eyeshines" recorded in annual spotlight surveys in 11 rivers. The line is a second order polynomial regression ($r^2$= 0.927, p= 0.0001).

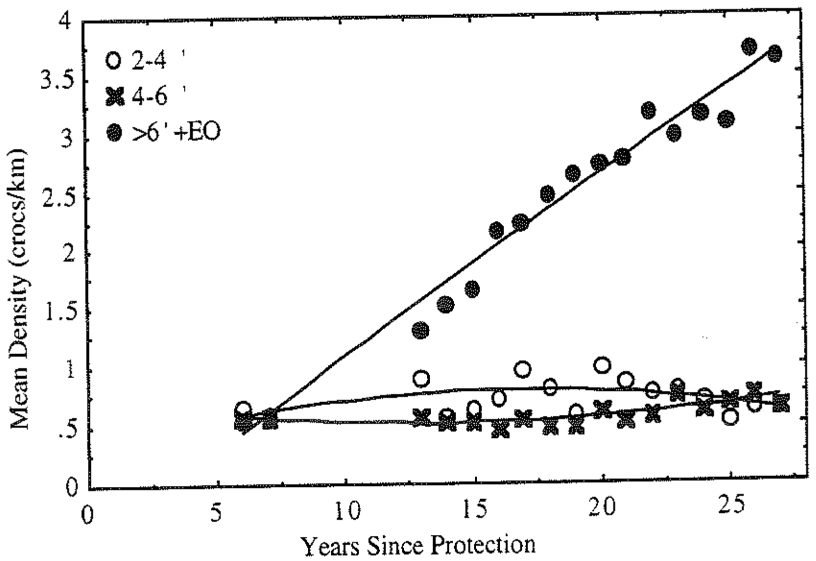


Figure 7. Mean density of different sized *C. porosus* sighted in spotlight surveys in 11 rivers (1977-98) as a function of years since protection (1= 1972; 27= 1998). Lines are linear and second order polynomial regressions indicating general trends.



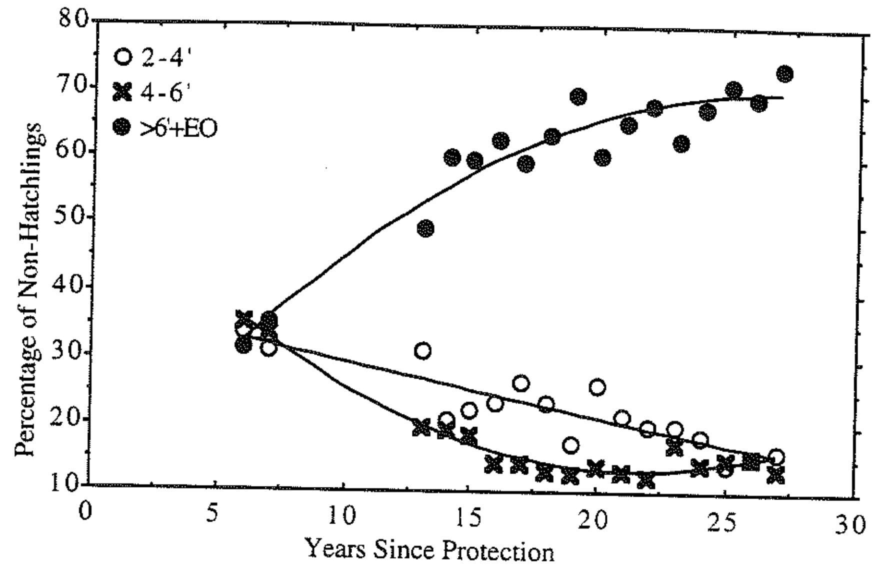


Figure 8. Percentage contribution of different size classes of *C. porosus* (2-4', 4-6' and >6'+EO) to the total number of non-hatchling crocodiles sighted in spotlight surveys in 11 rivers since protection (1= 1971; 27= 1998)

In 1977, the 2-4', 4-6' and >6'+EO components of the population had already reached a stage where they were contributing equally to the total non-hatchling crocodiles sighted (Fig. 8). By 1998, the two smaller size classes (2-4' and 4-6') together made up about 25% of all *C. porosus* sighted, and the larger animals (>6'+EO) 75%. The increase in the percentage of >6'+EO size class is highly significant ($r^2$= 0.93, p= 0.0001; Fig. 8), and better described by a polynomial than a linear model. The decline in the percentage composition of 2-4' crocodiles is linear ($r^2$= 0.77, p= 0.001), but a polynomial model is a better fit than a linear model to the trends in 4-6' crocodiles ($r^2$= 0.956, p= 0.0001), supporting the view that this size class may be reaching a stable percentage of the population.

The full extent of the recovery is perhaps more simply indicated by comparing the size structure of all *C. porosus* sighted in 1997 surveys in all rivers (N= 11) with the size structure of animals sighted in the first surveys carried out in all rivers surveyed in 1975-76 (N= 35 rivers; Table 3; Fig. 9).

That hatchlings make up similar percentages of all animals sighted in both the early and recent surveys, is in part a reflection of egg harvests introduced after 1983. However, that the relationship between hatchling numbers and the numbers of 2-3', 3-4' and 4-5' size categories has changed greatly is consistent with survival rates declining as the population recovered. The numbers of 5-6' crocodiles relative to the numbers of hatchlings has remained similar, suggesting that the increased losses of juveniles has a negligible effect on the overall population.

Changes over time in the adult segment of the population (>7' size class) can be examined by looking at 2'-long size categories for the years 1984-1997, where most consistent data were available (Table 4).

Table 3. Mean percentage of *C. porosus* in different size classes sighted in 1975-76 (35 rivers) and 1997 (11 rivers). <2' = hatchlings.

| Size Class (feet) | Mean Percentage 1975-76 | 1997 |
|---|---|---|
| <2' | 27.1 | 25.0 |
| 2-3' | 25.2 | 7.3 |
| 3-4' | 15.8 | 4.5 |
| 4-5' | 9.5 | 5.4 |
| 5-6' | 5.9 | 5.9 |
| 6-7' | 2.8 | 7.5 |
| >7' | 3.2 | 19.6 |
| EO | 10.0 | 24.7 |

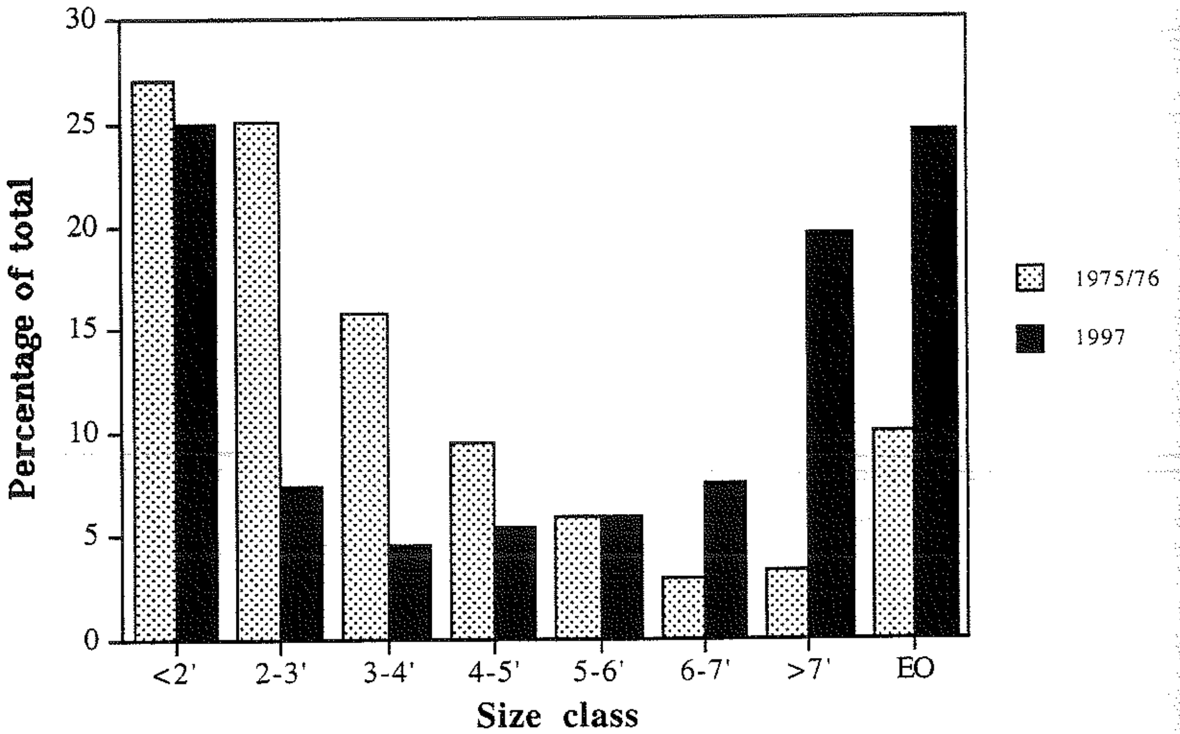


Figure 9. Population size structure of *C. porosus* recorded in spotlight surveys in 35 rivers in the Northern Territory in 1975/76 (shaded bars) and 11 rivers in 1997 (solid bars; Table 3).



Table 4. Mean density of 2' size classes (estimated total lengths) of *C. porosus* (>6') recorded during spotlight surveys between 1984 and 1997. "+/-" indicates an increasing or decreasing trend over time; $r^2$ and p refer to linear regressions over time; N= no. river units.

| Year | N | km | 6-8' | 8-10' | 10-12' | 12-14' | 14-16' | >16' |
|---|---|---|---|---|---|---|---|---|
| 1984 | 15 | 832 | 0.313 | 0.117 | 0.026 | 0.014 | 0.006 | 0.001 |
| 1985 | 15 | 523 | 0.388 | 0.164 | 0.061 | 0.011 | 0.004 | 0.000 |
| 1986 | 15 | 601 | 0.368 | 0.196 | 0.106 | 0.025 | 0.005 | 0.002 |
| 1987 | 15 | 788 | 0.463 | 0.326 | 0.127 | 0.048 | 0.009 | 0.000 |
| 1988 | 15 | 791 | 0.470 | 0.378 | 0.087 | 0.019 | 0.006 | 0.001 |
| 1989 | 11 | 520 | 0.383 | 0.267 | 0.063 | 0.008 | 0.004 | 0.000 |
| 1990 | 11 | 515 | 0.447 | 0.260 | 0.031 | 0.014 | 0.000 | 0.000 |
| 1991 | 11 | 508 | 0.409 | 0.309 | 0.118 | 0.047 | 0.010 | 0.000 |
| 1992 | 15 | 549 | 0.585 | 0.332 | 0.067 | 0.016 | 0.005 | 0.000 |
| 1993 | 15 | 560 | 0.614 | 0.370 | 0.082 | 0.029 | 0.005 | 0.000 |
| 1994 | 15 | 549 | 0.583 | 0.355 | 0.066 | 0.042 | 0.009 | 0.000 |
| 1995 | 15 | 778 | 0.704 | 0.356 | 0.090 | 0.045 | 0.013 | 0.000 |
| 1996 | 15 | 787 | 0.780 | 0.391 | 0.080 | 0.050 | 0.011 | 0.000 |
| 1997 | 15 | 786 | 0.908 | 0.450 | 0.089 | 0.043 | 0.015 | 0.000 |
| trend | | | + | + | | + | + | - |
| $r^2$ | | | 0.827 | 0.720 | n/s | 0.374 | 0.388 | 0.293 |
| p | | | 0.0001 | 0.0001 | | 0.02 | 0.02 | 0.046 |

Crocodiles greater than 12' have only ever been recorded at low densities, and those above 16' have always been rare. This partly reflects increased wariness of larger crocodiles with increasing size and age (Webb and Messel 1979), although it is clear than crocodiles over 16' have never been common, even in the early days of commercial hunting (Webb *et al.* 1984).

When densities are recalculated into three size classes (6-10', 10-12', >12'), significant increases in the density of 6-10' crocodiles ($r^2$= 0.86, p= 0.0001) and 12'+ crocodiles ($r^2$= 0.38, p= 0.018) are apparent, relative to stability in the 10-12' crocodiles (Figs. 10 and 11).

This appears to reflect sex-specific differences in growth rates and maximum size, which create bottlenecks. Wild male *C. porosus* mature at about 3.0-3.4 m total length (10-11'), which takes around 16 years, and then they continue to grow to a maximum length which normally ranges from 3.7 m (12') to over 5.2 m (17'). On average, most adult males never exceed 4.3 to 4.6 m (14-15') which is supported by the lack of crocodiles over 4.6 m (15') encountered historically (Webb *et al.* 1984). In contrast, wild female *C. porosus* mature earlier, between 2.1 m (7') and 2.3 m (7.5') and between 12 to 16 years of age, but maximum size lies in the range 3.0 m (10') to 3.4 m (11').

As relatively few females exceed 10', they accumulate in the 10-12' size class over time: a bottleneck of females. When 32 crocodiles greater than 7' were caught randomly from a 33 km stretch of the Adelaide River in 1994, 10 were greater than 10' and were all males. The remaining 22 between 7' and 10' comprised 21 females and 1 male.

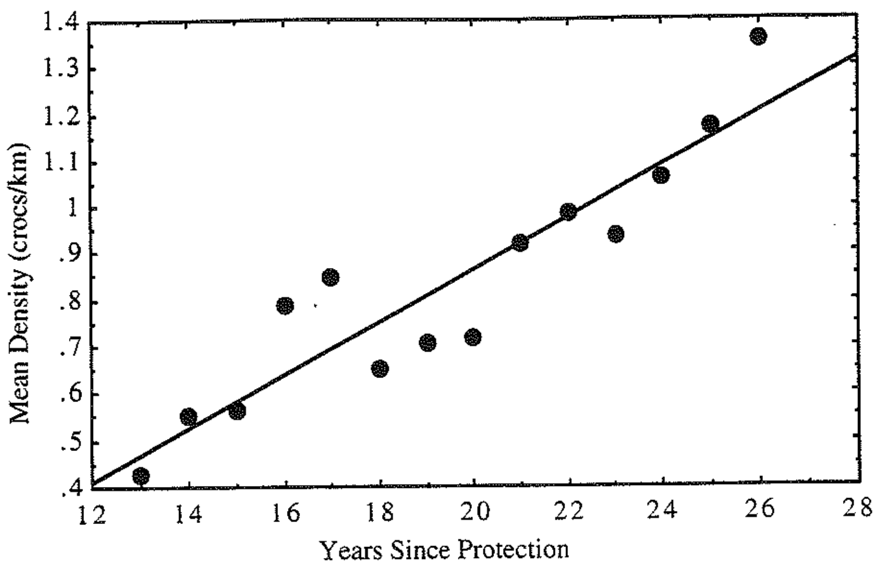


Figure 10. Mean density of 6-10' long *C. porosus* sighted in 11 major rivers systems in the Northern Territory; 1984-1997. The relationship is described by a linear regression. Years Since Protection: 1= 1972; 27= 1998.

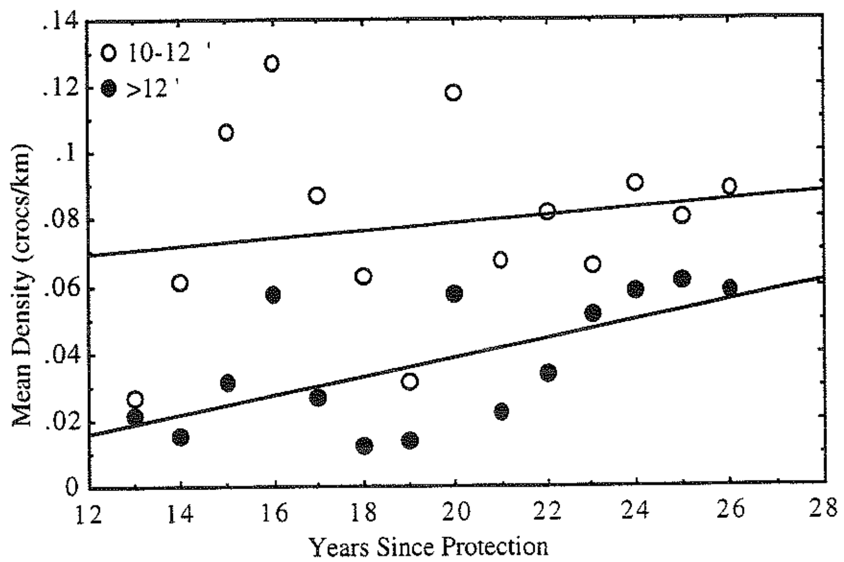


Figure 11. Mean density of 10-12' and >12' long *C. porosus* sighted in 11 major rivers systems in the Northern Territory, 1984-1997. Relationships are described by linear regressions. Years Since Protection: 1= 1972; 27= 1998.

The majority of individuals growing beyond this 10'-12' bottleneck are males, which are still short of their maximum size. Densities within the 10-12' size class have remained relatively constant, suggesting that the levels of male recruitment into the larger size classes (> 12') may be stable.



The male bottleneck does not occur until the 12-14' size range, and relatively few males pass beyond this into the >14' class. However, the results show that the >12' size class is continuing to increase significantly, indicating that the upper limit for adults has yet to be reached. In other words, the mean size of *C. porosus* in the Northern Territory is continuing to increase, despite densities probably approaching carrying capacity.

### 5.5. River-specific differences

The 11 rivers used for these analyses all contain medium to high densities of saltwater crocodiles, but the recovery occurred differently in different rivers, depending on the history of hunting and the extent to which habitats were intact. Two examples are given below, one in which eggs have been harvested continually since 1983 and one in which they have not.

*a. Adelaide-Mary System*

The adjacent Adelaide and Mary Rivers are two of the largest river systems in the Northern Territory. Both sites experienced intensive hunting pressure due to the high densities of *C. porosus* they contained and their close proximity to Darwin (Webb *et al.* 1984). Subsequent recovery has been important for crocodile ecotourism on the rivers, but has also resulted in greater predation on grazing livestock on adjacent lands. Moreover, the threat to people themselves has increased, particularly recreational fishermen, who also (and rightly) complain that crocodiles are taking important game fish species such as barramundi.

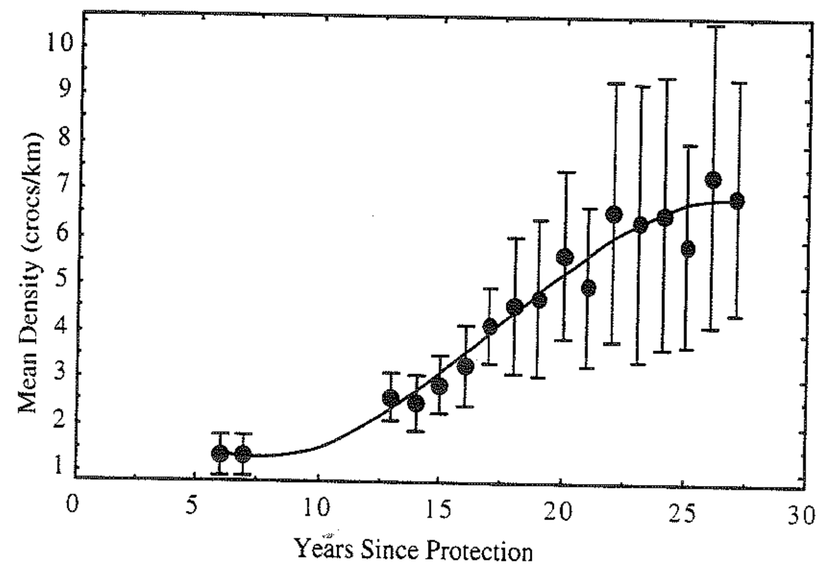


Figure 12. Mean density of non-hatchling *C. porosus* in the Adelaide and Mary River systems, as revealed by spotlight counts in 5 survey sections since 1977. Vertical bars are one standard error on each side of the mean, and the line is a third order polynomial regression (mean density= 4.663 + 0.983YSP + $0.083YSP^2$ - $0.002YSP^3$; $r^2$= 0.96, p= 0.0001). Years Since Protection (YSP): 1= 1972; 27= 1998.

In 1971, saltwater crocodiles were hard to find anywhere in the Adelaide and Mary River systems; they had been largely eradicated and the floodplain grasses used for riverside nesting had all but disappeared. So the build-up to over 1 crocodile per kilometre in 1977 itself represents a substantial recovery. The 4-5 times increase in density since 1977 (6 YSP; Fig. 12) now appears to be reaching a plateau in the Adelaide River, but continues to increase in a near linear fashion in some parts of the Mary River.

As mentioned above in connection with Figure 4, densities of saltwater crocodiles in the Sampan Creek-Alligator Lagoon section of the Mary River reached 15-20 crocodiles/km by the late 1990s, which matches early reports about this unusual "high-density" river from the 1940s. This is perhaps the highest concentration of wild *C. porosus* known from anywhere today, yet nesting in the Mary River was almost non-existent in the 1970s and even today is very poor: perhaps 10-20 nests in the whole system. Recruitment has taken place by immigration. This survey section is renowned for its abundant fish stocks, and it seems likely that food availability and crocodile density are linked.

There has been a pronounced shift from small to large crocodiles since the late 1970s (Fig. 13) in these two systems, with the largest size class (>6'+EO) showing a highly significant increase over time ($r^2$= 0.97, p= 0.0001, third order polynomial), which is not declining. The intermediate 4-6' size class has also shown a significant increase over time ($r^2$= 0.61, p= 0.0054; third order polynomial), which began 16 YSP and has seen densities double in the last 6 years. The smallest size class (2-4') has shown a threefold increase in density since 1977 ($r^2$= 0.66, p= 0.0022 third order polynomial), but stabilised and is possibly declining after 21 YSP. The percentage composition of the population (Fig. 14) also clearly shows this shift to larger animals, which by the late 1990s comprise over two thirds of animals sighted in surveys.

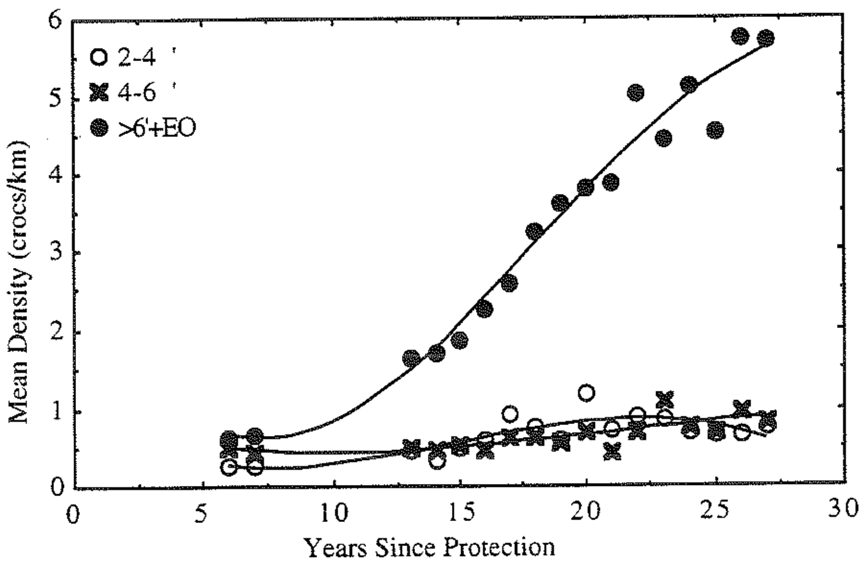


Figure 13. Mean density of non-hatchling *C. porosus* (2-4', 4-6' and >6'+EO) sighted over time in the Adelaide and Mary River systems combined. Relationships are described by third order polynomial regressions. Years Since Protection: 1= 1972; 27= 1998.



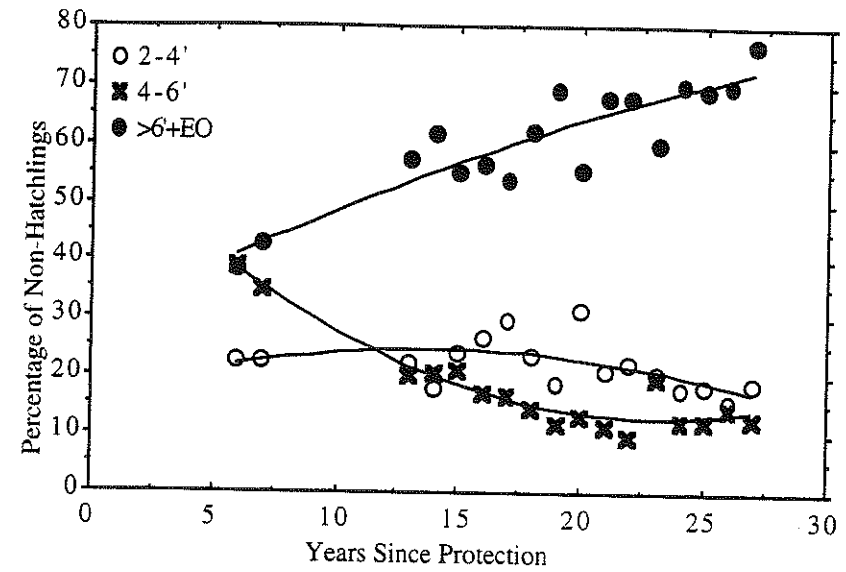


Figure 14. Percentage contribution of three size classes (2-4', 4-6', >6'+EO) to the total number of non-hatchling *C. porosus* sighted against years since protection in the Adelaide and Mary River systems combined. Relationships are second order polynomial regressions. Years Since Protection: 1= 1972; 27= 1998.

*b. Kakadu System*

The four major river systems associated with Kakadu National Park (East Alligator, South Alligator, West Alligator, Wildman) were not included in the nest harvesting program initiated in 1983 (12 YSP).

Like the Adelaide and Mary Rivers, the Alligator Rivers were subjected to heavy harvesting before 1971, but nesting habitats were badly damaged by overgrazing. However, the Alligator Rivers did contain extensive freshwater swamps and billabongs in which remnant populations of saltwater crocodiles existed at the time of protection, and the recovery of this population was largely due to local breeding. Between protection and 1977 the population increased significantly, and since that time (Fig. 15) it has increased by a factor of 2.4. However, from the 1980s onward, densities have tended to stabilise.

Like the Adelaide-Mary system described above, there has been a shift from small to large size classes of *C. porosus*, with a highest rate of increase seen in >6' animals ($r^2$= 0.903, p= 0.0001; second order polynomial) (Fig. 16). The intermediate 4-6' size class has shown a slight increasing trend ($r^2$= 0.40, *p*= 0.028, second order polynomial) whereas the 2-4' size class increased until around 15 YSP and then declined ($r^2$= 0.41, p= 0.026, second order polynomial). The >6'+EO size class increased from 40% to 70% of the animals sighted in spotlight surveys between 1977 and 1998 and still appears to be increasing ($r^2$= 0.75, p= 0.0001, second order polynomial). The percentage contribution of different size classes has been reasonably stable since 1992 (Fig. 17).

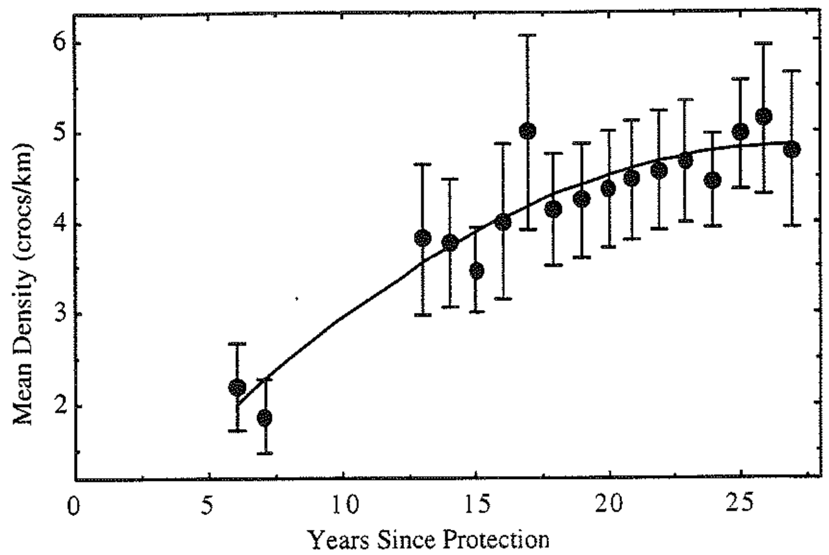


Figure 15. Mean density of non-hatchling *C. porosus* sighted in spotlight surveys in the Wildman, West Alligator, South Alligator and East Alligator River systems. Vertical bars represent one standard error around the mean and the relationship is a second order polynomial regression (mean density= 0.174 + 0.338YSP - 0.006YSP$^2$; $r^2$= 0.89, p= 0.0001). Years Since Protection (YSP): 1= 1972; 27= 1998.

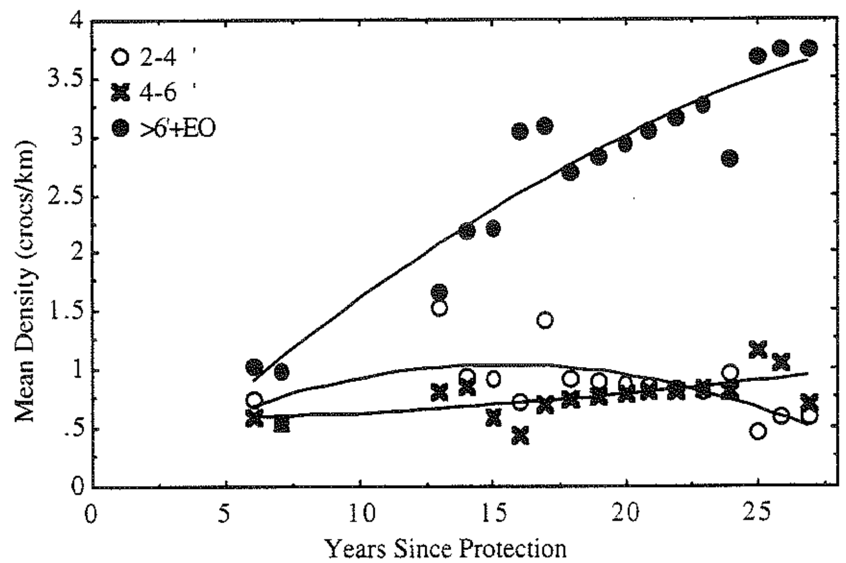


Figure 16. Mean density of three size classes (2-4', 4-6' and >6'+EO) of non-hatchling *C. porosus* sighted in spotlight counts in the Wildman, West Alligator, South Alligator and East Alligator Rivers combined. Relationships are second order polynomial regressions. Years Since Protection: 1= 1972; 27= 1998.



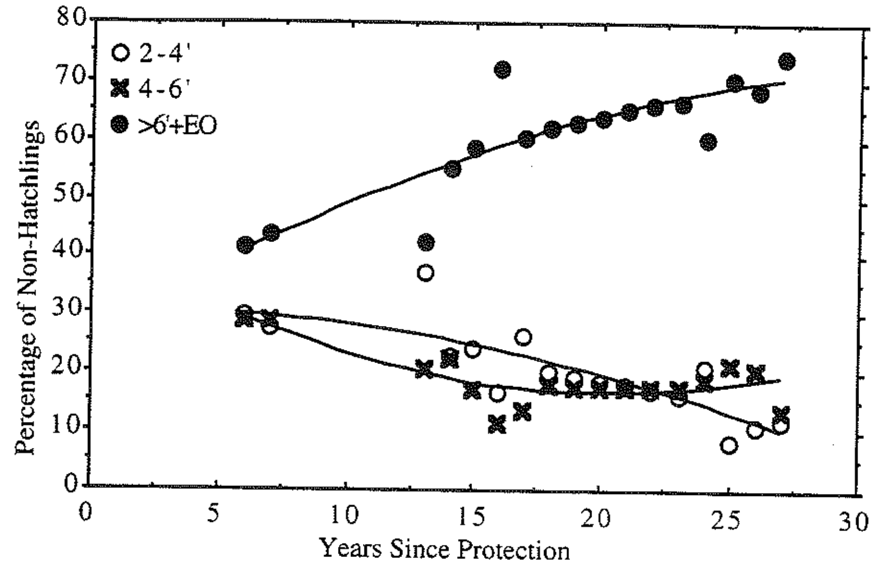


Figure 17. Percentage contribution of three size classes (2-4', 4-6', >6'+EO) to the total numbers of non-hatchling *C. porosus* counted in spotlight surveys in the Wildman, West Alligator, South Alligator and East Alligator Rivers. Relationships are second order polynomial regressions. Years Since Protection: 1= 1972; 27= 1998.

### 5.6. Overall Trends in Spotlight Count Results

From spotlight survey data since 1977, four main trends in recovery in different river systems can be identified.

Trend 1. Linear increases in non-hatchling density from 1977 onward, which in some rivers spans a 24 year period (Blyth, sections of the Mary, Reynolds mainstream, South Alligator and Wildman Rivers). High numbers of juveniles were already established by 1977, and the ongoing increase in density was associated with a shift from small to larger animals. Carrying capacity does not appear to have been exceeded. Some systems (eg Mary River) have low recruitment from nesting but high recruitment from immigration.

Trend 2. Linear population increases after 1977, but with a greatly reduced rate of increase by 1998 (eg Daly, Adelaide, East Alligator, Glyde, Liverpool, Tomkinson and West Alligator Rivers). Juveniles were well established by 1977 and the potential for ongoing breeding was relatively high. There was typically a rapid increase in density after 1977 until the capacity to accommodate further expansion became compromised, leading to a plateau. Social exclusion and cannibalism (Webb and Manolis 1991) could both be involved.

Trend 3. No increase or decrease in counts during the period of surveys (eg Cadell, Finniss mainstream, Reynolds freshwater lagoons and Wurugoij Rivers). The extent of recovery up to 1977 was such that no carrying capacity appears to have been reached. The number of animals sighted each year was highly variable, but no significant trends were apparent over the survey period.

Trend 4. Decrease in crocodile counts after 1977. Relatively few areas show sustained decreases. *C. johnstoni* numbers in the upstream parts of the Adelaide declined, probably in response to increasing *C. porosus* numbers. Some declines in minor waterways and sidecreeks, which only ever contained low densities, may reflect larger crocodiles taking up residence which are less tolerant of conspecifics. In some billabongs (Sweets Billabong in the Finniss River), dense aquatic plants have re-established over time, such that reductions in the numbers of crocodiles counted reflect visibility biases rather than real changes in population size.

## 6. THE TOTAL POPULATION (1977-98)

The 11 rivers discussed above reflect medium to high density situations, and often rivers in which breeding takes place. They are sites which are generally net providers of crocodiles to NT wetlands, although this is not the case with the Mary River. The majority of NT wetlands have limited breeding and only ever contained low densities of crocodiles, mainly received through emigration from breeding areas. Thus trends in the wild population as a whole cannot be assessed from the results above. That is, it is unclear whether crocodiles being lost from the major breeding rivers were themselves building up in the population outside those rivers.

### 6.1. Analysis Methods

In 1989, a helicopter survey program which sampled one or two 10 km long survey segments in 68 rivers was introduced (Webb and Manolis 1991) to monitor whether the total population was increasing, decreasing or stable. This broad-brush approach was not designed to monitor the status of crocodiles in particular rivers (as discussed above), but rather to monitor trends in the total population. In 1997, a subsample of 21 sample units was selected which gave equivalent accuracy and precision (Britton *et al.* 1998).

### 6.2. Density

The general trends (Fig. 18) in mean densities derived from helicopter surveys (1989-98) and from historical spotlight data converted to helicopter count equivalents (1975-88), parallel those from the spotlight counts in the 11 major rivers (Fig. 5). [This is not surprising given that historical data converted from spotlight counts to helicopter shows less variability than would have been expected from real helicopter counts for those years].

These results suggest a Trend 2 recovery pattern was characterising the population as a whole after the basic increases which occurred between protection and 1977: a linear increase from mid-1970s to the early 1990s, followed by a plateau or stabilisation of numbers.

Low densities reported in the 1994 and 1998 were likely due to observer biases, because different spotters were used compared with all other years since 1989. However, helicopter counts are reasonably precise (Bayliss *et al.* 1986), and the results do confirm that the wild population as a whole was not absorbing or at least retaining crocodiles moving out of the major breeding areas.



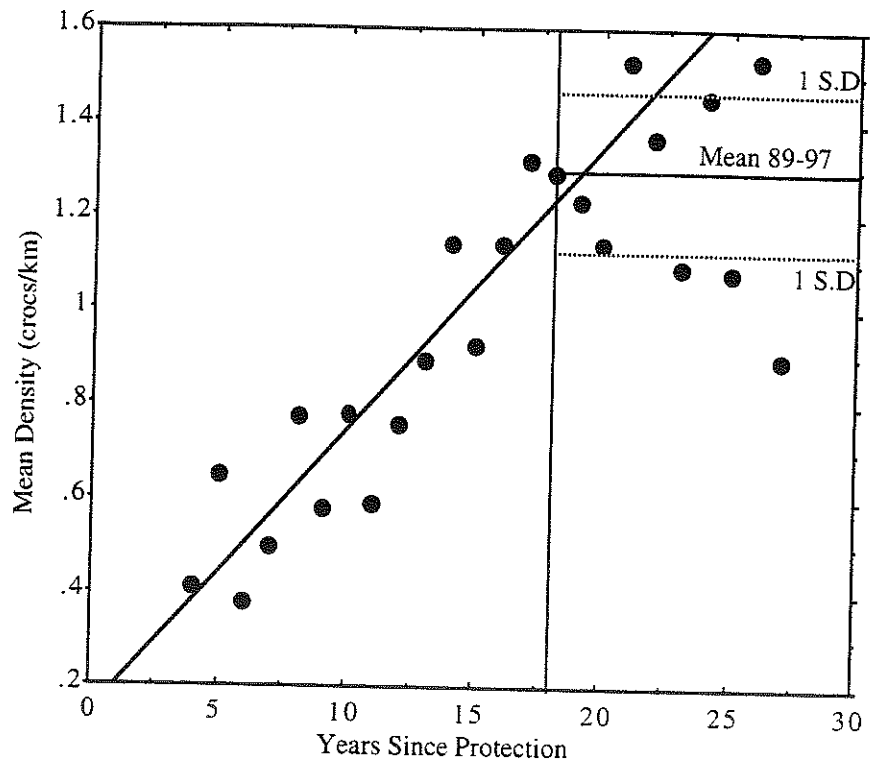


Figure 18. Mean density of *C. porosus* as indicated by helicopter surveys in 21 sample areas. Values to the left of the vertical line (before 1989, 18 YSP) are helicopter count equivalents predicted from spotlight counts. The horizontal line is the mean density between 1989 and 1997 (there was no significant trend) with one standard deviation above and below the mean (dotted line). The bold line is the linear regression between 1975 and 1993. Years Since Protection (YSP): 1= 1972; 27= 1998.

### 6.3. Population Structure

The size structure of crocodiles seen in the helicopter surveys (Table 5) was compared with size structure analysis from the spotlight surveys in 11 rivers (Table 1): only data for 21 helicopter units surveyed were used (1989-98) (Table 5, Fig. 19).

The 2-4' size class demonstrated a significant decline ($r^2$= 0.65, p= 0.005; linear regression) over the survey period, consistent with general trends in the spotlight count data for 2-4' crocodiles after 18 YSP (Fig. 7). Neither of the other two size classes showed any significant trend. When the 4-7' and >7' size classes are grouped (>4' crocodiles), the helicopter surveys indicate a significant increase in the density of larger crocodiles over time ($r^2$= 0.49; p= 0.02; linear regression), which is consistent with the general trends from the spotlight counts for 4-6' and >6'+EO crocodiles after 18 YSP (Fig. 7).

Table 5. Size structure of *C. porosus* as revealed through helicopter counts in 21 survey units over the period 1989 to 1998. N= total number sighted; D= density over the complete survey area; %= percentage composition of total numbers (non-hatchlings).

| Year | Area (km) | Total | | Small (2-4') | | Medium (4-7') | | Large (7-11') | | Extra-large (>11') | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | N | D | % | D | % | D | % | D | % | D |
| 1989 | 210 | 271 | 1.290 | 7.7 | 0.100 | 24.4 | 0.314 | 46.9 | 0.605 | 21.0 | 0.271 |
| 1990 | 210 | 259 | 1.233 | 10.0 | 0.124 | 22.8 | 0.281 | 54.1 | 0.667 | 12.7 | 0.157 |
| 1991 | 210 | 240 | 1.143 | 9.2 | 0.105 | 26.3 | 0.300 | 44.2 | 0.505 | 16.7 | 0.190 |
| 1992 | 210 | 321 | 1.529 | 9.3 | 0.143 | 37.4 | 0.571 | 40.5 | 0.619 | 11.8 | 0.181 |
| 1993 | 210 | 287 | 1.367 | 3.1 | 0.043 | 24.7 | 0.338 | 55.1 | 0.752 | 16.7 | 0.229 |
| 1994 | 210 | 229 | 1.090 | 5.2 | 0.057 | 32.8 | 0.357 | 39.3 | 0.429 | 22.7 | 0.248 |
| 1995 | 210 | 305 | 1.452 | 3.3 | 0.048 | 27.5 | 0.400 | 50.5 | 0.733 | 19.7 | 0.286 |
| 1996 | 210 | 227 | 1.081 | 4.4 | 0.048 | 25.1 | 0.271 | 50.7 | 0.548 | 19.4 | 0.210 |
| 1997 | 210 | 322 | 1.533 | 5.0 | 0.076 | 28.6 | 0.438 | 45.7 | 0.700 | 18.9 | 0.290 |
| 1998 | 210 | 199 | 0.948 | 1.5 | 0.014 | 22.1 | 0.210 | 53.8 | 0.510 | 22.6 | 0.214 |

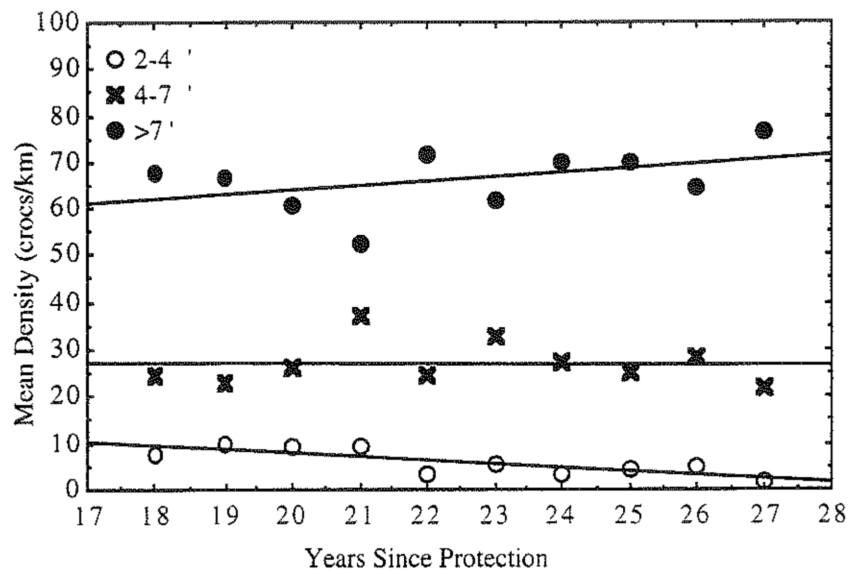


Figure 19. Percentage contribution of different size classes of *C. porosus* (2-4', 4-7' and >7') recorded during helicopter surveys in 21 survey units in the Northern Territory between 1989 (18 YSP) and 1998 (27 YSP).

### 6.4. Summary

That the spotlight counts in the 11 tidal rivers indicated a small but highly significant increase in mean density ($r^2$= 0.72, p= 0.002; linear regression) over the period 1989-98, whereas helicopter counts indicated no significant change [($r^2$= 0.03; p= 0.65): mean= 1.27 ± 0.21 (SD) crocodiles sighted per kilometre], is consistent with emigration from the major



rivers having little real impact on the low densities of crocodiles occupying most NT wetlands and comprising much of the NT population as a whole.

Crocodiles could be moving through to Queensland or Western Australia, or even to Indonesia (crocodiles have been sighted swimming past oil rigs in the Timor Sea), but are clearly not building in numbers commensurate with the numbers being lost from the major breeding rivers. As suggested by Messel *et al.* (1981), they could be experiencing much greater mortality rates once they leave rivers and start moving around the coast.

## 7. ESTIMATES OF THE TOTAL WILD POPULATION IN THE NORTHERN TERRITORY

Spotlight and helicopter counts provide density indices in areas surveyed, yet a wide variety of wetland habitats outside of areas are occupied by *C. porosus* but impossible to survey with these methods. Even for areas accessible to survey, correction factors enabling absolute abundance to be derived from measures of relative abundance need to be derived and all have errors associated with them (Messel *et al.* 1981a; Webb *et al.* 1984, 1989, 1990a; Bayliss *et al.* 1986; Bayliss 1987).

Attempts to estimate total population size in the early 1980s assumed that the majority of non-hatchling crocodiles were found in tidal rivers (Webb 1978, Messel *et al.* 1981a), but later research (Webb *et al.* 1983f), aerial surveys, and interviews with former hunters (Webb *et al.* 1984) revealed that the population outside of tidal rivers was perhaps half the total population (Webb *et al.* 1984). After quantifying the extent of available habitat, deriving estimates of absolute density for each habitat type, and making a series of reasonably conservative assumptions and corrections, an estimate of approximately 40,000 non-hatchling saltwater crocodiles was obtained for 1984 (Webb *et al.* 1984). The mean spotlight and helicopter count trends (Figs. 5 and 18) indicate an increase of 72-73% in non-hatchling densities between 1984 and 1998, suggesting the total population in 1998 would be 68,000 to 69,200. However, of the many possible errors and biases involved one is very significant. The proportion of non-hatchlings sighted in surveys decreases with increasing size (Webb *et al.* 1984; Bayliss *et al.* 1986). Thus, as the mean size of crocodiles in the population has increased since 1984, so the percentage seen in surveys has been reduced. The most realistic estimate for the current population is conservatively 70,000 to 75,000 non-hatchlings.

## 8. TRENDS IN NESTING AS INDICATORS OF POPULATION RECOVERY

Extensive research into nesting biology and habitats in the Northern Territory began in the 1970s and was stimulated by the introduction of ranching in the 1980s [eg Webb 1977a; Webb *et al.* 1977; Magnusson 1979a, 1979b, 1980a, 1980b, 1981, 1982; Magnusson and Taylor 1980; Magnusson *et al.* 1978a, 1978b, 1980; Messel *et al.* (Monographs 1-19)]. Yet the systematic monitoring of nest numbers is logistically difficult and was only a primary goal at one site (Melacca Swamp on the Adelaide River). In some areas where eggs were collected (eg Adelaide River, Finniss and Reynolds River systems) a reasonably systematic procedure was adopted from year to year, using largely the same observers, so some reasonable data on trends are available. However, with nesting in the wet season, spread over a 6-7 month period (November to April-May), many nests being in heavily vegetated freshwater swamps, and all nests subject to flooding, the logistics of nest surveying are great.

In 1979, eight years after protection, a helicopter survey over the complete length of the Adelaide River in the peak of the nesting season (February) revealed 2 nests beside the mainstream and eight nests in Melacca Swamp (which drains into the Adelaide River). From the 1979/80 wet season onward, surveys in Melacca Swamp (Fig. 20) identified most nests (over 90% per year). There was a significant increase in the number of nests over the entire survey period ($r^2$= 0.42, p= 0.003; linear regression) but due totally to a large and sustained increase after the 1994/95 season. Between 1979/80 and 1994/95 (9 to 24 YSP) nest numbers fluctuated around a stable mean. The four sharp declines in nesting (1979/80, 1982/83, 1987/88 and 1990/91 seasons) are very real trends and are not survey biases. It indicates that nesting effort in the same area, with a similar population being resident, can be reduced by 50% or more from year to year, due largely to climate (water level changes).

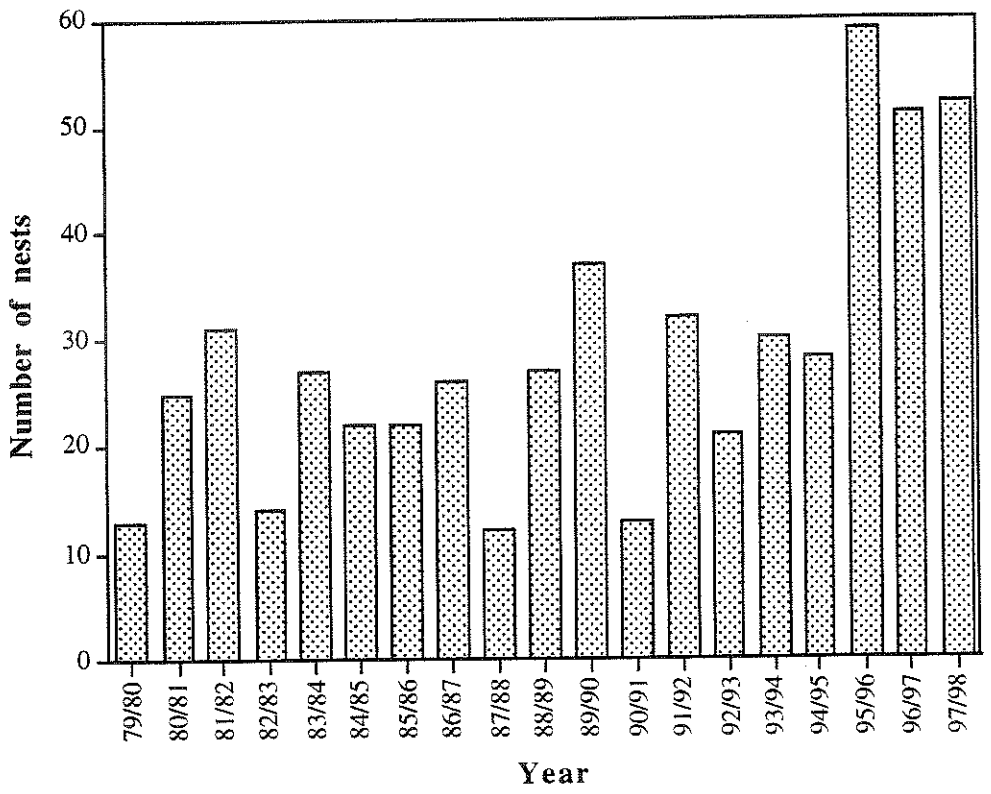


Figure 20. Numbers of *C. porosus* nests located in Melacca Swamp in each nesting season since the 1979/80 wet season, using similar search effort over the same areas.

The marked increase in nesting from 1995/96 (Fig. 20) onward correlates with a sharp increase in the density of 6-8' *C. porosus* recorded in spotlight surveys of tidal rivers (Table 4), and it seems likely that more individuals are moving into Melacca Swamp to nest. Between 1979 and 1994 the number of nests in Melacca was completely independent of 6-8' crocodile density in nearby tidal rivers ($r^2$= 0.002, p= 0.85), suggesting that it was a somewhat isolated enclave of adults which had survived the hunting period.



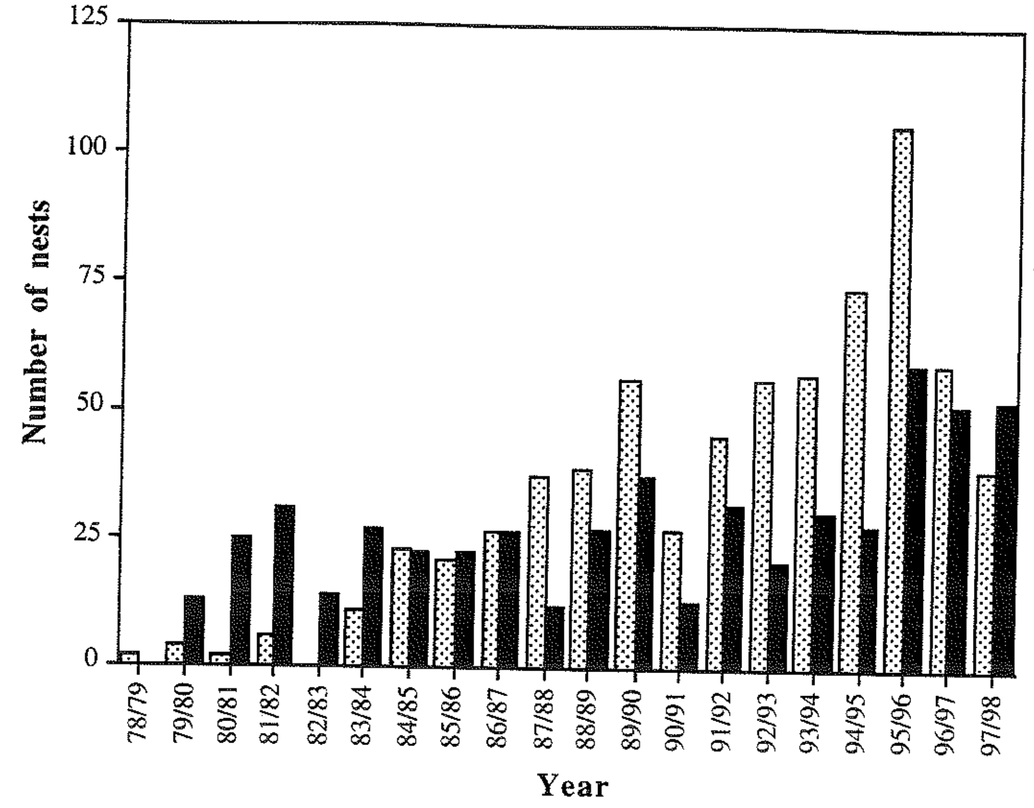


Figure 21. Number of *C. porosus* nests (excluding false nests) located in the Adelaide River system (shaded bars) and Melacca Swamp (solid bars). Data for 1997/98 are incomplete for the Adelaide River.

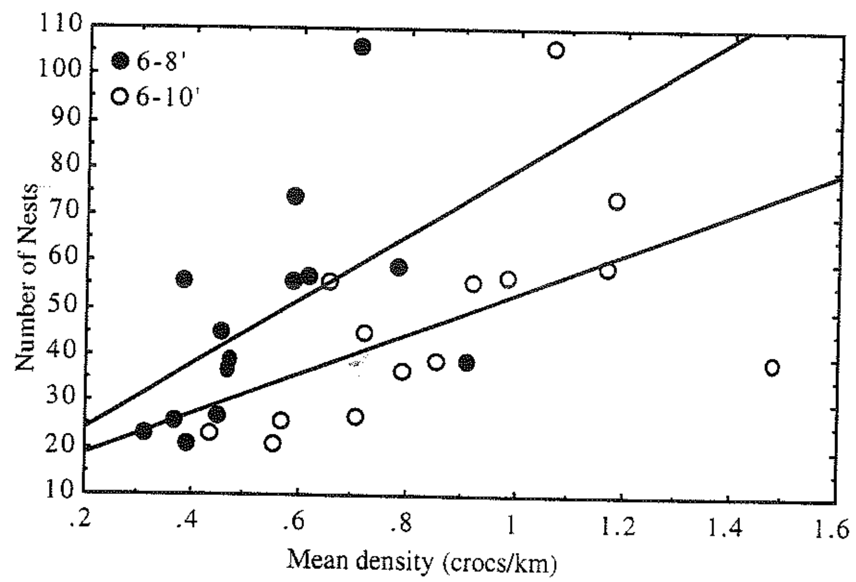


Figure 22. Relationship between numbers of *C. porosus* nests located in the Adelaide River and the mean density of 6-8' and 6-10' crocodiles recorded from spotlight surveys in tidal rivers generally undertaken in the preceding dry season (1984/85 to 1996/97 seasons).

Along the Adelaide River itself, a completely different pattern of nesting was recorded (Fig. 21). Notwithstanding biases due to less rigorous nest searching (particularly in the 1996/97 and 1997/98 seasons), and extensive flooding (1996) which destroyed significant habitat used for nesting up until that time, around 80% of nests were probably detected. The adults in this river had been nearly completely removed prior to 1971, and from a baseline of 2 nests located 8 years after protection (1979), nesting increased to 12 nests by 1983/84, and over the next 12 years (to 1994/95), increased at a mean rate of 4.6 nests (NN) per year (NN= -41.8 + 4.64YSP; $r^2$= 0.80, p= 0.0001; linear regression). This increase was strikingly apparent along the long floodplain meanders of the Adelaide River, where the ability to detect nests is very high, and is not a bias associated with the odd new patch of nesting habitat being found. Unlike Melacca Swamp, the increase in nesting along the Adelaide River between 1983/84 and 1995/96 was highly correlated with the increasing density of 6-8' crocodiles sighted in spotlight counts in tidal rivers (Table 4) ($r^2$= 0.58, p= 0.003; linear regression), and was even more highly correlated with the density of 6-10' crocodiles sighted ($r^2$= 0.60, p= 0.002; linear regression)(Fig. 22).

In the Finniss-Reynolds River system, the third area for which considerable data have been collected (Fig. 23), there are extensive freshwater swamps which support nesting and which provided a refuge for some adults during the hunting period (Webb *et al.* 1984). Search effort varied over time, and no data are available for 1996/97 and 1997/98: the recent reduction in nest numbers on the Reynolds River was either due to flooding (1994/95) or missing data (1996/97, 1997/98). Nest numbers have clearly increased over time but then stabilised.

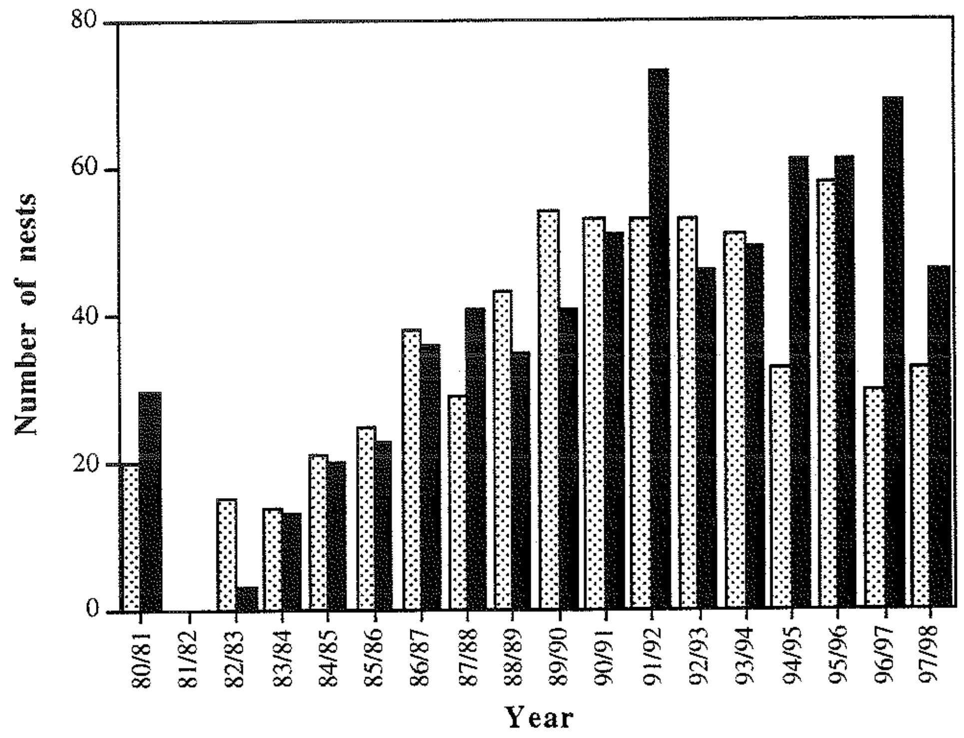


Figure 23. Numbers of *C. porosus* nests located in the Reynolds (shaded bars) and Finniss (solid bars) River systems over time. Data for 1996/97 and 1997/98 are incomplete.



The numbers of nests found in the Finniss River and the combined Finniss-Reynolds system are correlated with the density of 6-8' and 6-10' crocodiles seen in spotlight surveys in Northern Territory rivers generally (Fig. 24; Table 6).

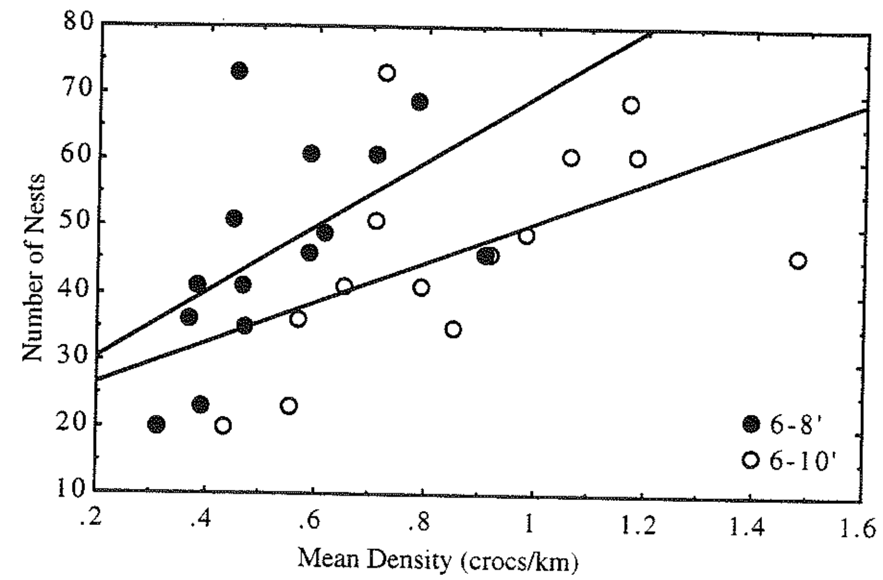


Figure 24. Relationship between numbers of *C. porosus* nests found in the Finniss River between 1984/85 and 1995/96 and the mean density of 6-8' and 6-10' crocodiles recorded from Northern Territory rivers in the spotlight survey program in the dry season preceding each nesting season. Lines are linear regressions (see Table 6).

Table 6. Levels of significance (p) and $r^2$ values for the linear regression relationships between the number of nests recorded in the Reynolds, Finniss and combined Reynolds-Finniss Rivers from 1984/85 to 1995/96, and the densities of 6-8' and 6-10' crocodiles in spotlight counts in major Northern Territory rivers (Fig. 24).

| | 6-8' | | 6-10' | |
|---|---|---|---|---|
| | $r^2$ | p | $r^2$ | p |
| Reynolds | 0.29 | 0.074 | 0.17 | 0.183 |
| Finniss | 0.39 | 0.029 | 0.43 | 0.020 |
| Combined | 0.41 | 0.025 | 0.36 | 0.040 |

## 9. IMPACTS OF HARVEST

As part of a sustainable use initiative to encourage conservation of crocodiles and nesting habitats in the Northern Territory, a ranching program was introduced (1983/84) which saw eggs harvested directly from the wild. Additional harvest of wild crocodiles has occurred in the form of a problem crocodile removal program, limited trial harvests of adults and subadults from the wild, hunting by Aboriginal people for eggs and crocodiles, and losses of crocodiles drowned in coastal fishing operations. Illegal hunting has occurred from time to time, but only at low levels.

Although basing the harvest on computerised simulation models was considered possible (eg Webb *et al.* 1984; Webb and Smith 1987), it was ultimately rejected because critical factors (eg age- and size-specific mortality rates and density dependent compensatory factors) could not be estimated accurately. Instead, it was decided to implement conservative harvests (rather than aim for a maximum sustainable yields), and to maintain long-term monitoring programs to determine whether the population rate of increase was greatly compromised.

The first phase of the ranching program was restricted to a relatively small number of rivers (Adelaide, Melacca, Finniss and Reynolds), but within these rivers the harvest was maximised. It soon became apparent that some undetected nests still hatched and that recruitment of one-year-olds into the population was not significantly different to those areas not subject to harvest (Webb *et al.* 1989): hatchling survival to one year of age appears density-dependent (Webb and Manolis 1991).

Table 7. *Crocodylus porosus* egg harvest data for the Northern Territory. *= underestimate of the total number collected because data are incomplete [1980/81, 1982/83, 1983/84, 1996/97, 1997/98]. ???= insufficient data provided [1997/98]

| Season | No. of Rivers | Total Eggs | Viable Eggs |
|---|---|---|---|
| 1978/79 | | 0 | 0 |
| 1979/80 | | 135 | 0 |
| 1980/81 | | 2,758 * | 100 |
| 1981/82 | | 327 | 0 |
| 1982/83 | | 298 * | 85 |
| 1983/84 | 5 | 2,320 * | 1,354 |
| 1984/85 | 4 | 3,518 | 2,493 |
| 1985/86 | 4 | 3,737 | 2,236 |
| 1986/87 | 5 | 4,401 | 2,760 |
| 1987/88 | 6 | 5,300 | 3,410 |
| 1988/89 | 11 | 6,497 | 3,886 |
| 1989/90 | 15 | 12,010 | 8,859 |
| 1990/91 | 19 | 9,212 | 5,491 |
| 1991/92 | 22 | 15,298 | 9,919 |
| 1992/93 | 23 | 12,379 | 8,538 |
| 1993/94 | 24 | 17,322 | 12,881 |
| 1994/95 | 24 | 19,033 | 13,106 |
| 1995/96 | 26 | 29,044 | 21,872 |
| 1996/97 | 26 | 19,494 * | 13,820 * |
| 1997/98 | 4 | 5,805 * | ??? |
| Totals | | 165,370 | 110,810 |

Over time, the egg harvest program was expanded throughout the Northern Territory, with larger numbers of eggs being harvested in the 1990s (29,000 eggs at its peak; Table 7), but with no return of raised animals back to the wild to compensate for the harvest. That this



level of harvesting is being sustained with no detectable effect on wild populations (Figs. 3, 21 and 23) is consistent with the rate of harvest being well below the intrinsic rate of increase in the wild populations.

It could be argued that the impacts of harvesting eggs would not be detected until years later, and that the current change in population structure generally (more large and less small crocodiles) is a reflection of the prolonged period of egg harvest. However, trends in harvested areas like the Adelaide-Mary (Figs. 12 and 13) are the same as those in non-harvested areas such as the Kakadu Rivers (Figs. 15 and 16). Thus although the harvest program no doubt reduces the numbers of hatchlings entering the wild population, it may have had a minimal impact on the numbers of larger crocodiles that ultimately enter the population.

## 10. TRENDS IN MANAGEMENT

Within the Northern Territory, and throughout their global range, saltwater crocodiles have had their populations greatly reduced over time. Excessive hunting for skins has been a significant factor in most areas, and was the prime reason behind population declines in the Northern Territory between 1945 and 1971. The wetland habitats within the Northern Territory are still largely intact, which is the exception rather than the rule throughout most of the countries within the range of *C. porosus*.

The goal of introducing protection in the Northern Territory (1971) was to remove the hunting pressure on *C. porosus*, and encourage a recovery in the wild populations. Due to the major research program on *C. porosus* instigated in the early 1970s by Professor Harry Messel, the early years of recovery were reasonably well documented. In addition, this program led to a vast amount of research information on saltwater crocodiles being gathered in a short period of time, and led to the development of a range of hypotheses about population dynamics and density-dependent influences on them. That the Federal Government and NT Governments both made a significant long-term commitment to maintaining crocodile survey programs, has meant that despite a varied history of management, monitoring has continued for over one quarter of a century.

As crocodiles started to recover in the Northern Territory, the public which had supported protection started to question the wisdom of their choice. Saltwater crocodiles are large and dangerous predators, and reintroducing them in all coastal wetlands meant that human safety would be increasingly compromised. By 1980, when the recovery was well underway, calls for culling the wild population became widespread.

The introduction of management programs aimed at increasing the value of *C. porosus* in the eyes of the public, although highly controversial at the time, played a significant role in changing the public perception of crocodiles from a liability to an asset. The promotion of tourism in the Northern Territory in the late 1970s furthered the value of crocodiles generally. The expanding populations of saltwater crocodiles were initially considered a threat to the embryonic tourist industry (1979/80), but crocodiles soon became a major focus of media attention, which indirectly promoted the Northern Territory as a tourist destination. As wild and captive crocodilians became primary tourist attractions, generating both wealth and employment, so the value of crocodiles to the eyes of the community increased.

The successful proposal to CITES in 1985 opened the door for ranching and for the production and sale of crocodile skins and meat. The highly controversial decision (1982) to allow landowners to levy a fee for freshwater crocodile eggs collected from their lands, was extended to the *C. porosus* ranching program, and ensured landowners were beneficiaries of this management program from the start, although their gains have often been minor.

When skins started to be exported (1987), there was increased pressure on Government to expand the crocodile industry and to increase the access by crocodile farms to wild crocodile resources. Crocodile conservation and management in the Northern Territory became far more politically sensitive than had been the case previously. The industry was expanded slowly, the ranching program was expanded according to previous plans, and access to wild crocodiles remained restricted with the exception of problem crocodiles.

The recovery of the wild populations continued, and the Northern Territory acquired a significant national and international reputation as a consequence. With ongoing research and management strategies promoting sustainable use, the program was often interpreted as a sound model of pragmatic, effective conservation.

In the late 1980s, support for crocodile research on crocodiles started to be withdrawn. The commitment by Government to crocodiles became more tightly focussed on meeting its management obligations under CITES alone, and supporting the crocodile farming industry. Responsibility for crocodile management, previously held by the wildlife authority, was subdivided between departments responsible for wildlife and conservation, primary production and industry development.

In the 1990s a series of support services (eg management of the ranching program, problem crocodile program) were privatised, and services previously contracted out (monitoring, analysis of results, reporting on the egg harvest) were absorbed by Government as in-house functions. The problem crocodile program was later reabsorbed by Government as an in-house operation. Trial harvests of some wild animals were initiated, with landowners being beneficiaries.

While these changes in management have taken place the wild populations of *C. porosus* have continued to expand. Although the population of non-hatchlings appears to be stabilising, the size structure of crocodiles greater than 6' in length continues to change in the direction of there being more large crocodiles.

## 11. CONCLUSIONS

The monitoring programs carried out in the Northern Territory from the early 1970s until the late 1990s have documented a very significant recovery in the Saltwater Crocodile population. It is now very clear that the small remnant populations which remained in 1971 (perhaps 5% of historical numbers), were more than enough to stimulate the recovery given habitats that were still largely intact. When seen from a different perspective, unrestricted harvesting for some 26 years did not compromise the ability of this species to recover if given the opportunity.

After a lag time of at least 10 years, marked increases in nesting occurred, although the contribution of increased recruitment through increased nesting was continually compromised by density-dependent factors. That is, initial recovery rates were fast because densities were low and the wild population is unlikely to expand more rapidly regardless of how many nests are now made annually.

The current wild population is estimated as 70,000 and 75,000 wild non-hatchling saltwater crocodiles, that occur in high densities in a few rivers and at low densities throughout the NT. The population is considered very close to the historical population in 1945, because



densities reported in different rivers today are similar to those reported in 1945 from a detailed historical review (Webb *et al.* 1984). The stability of most population indices today suggests that carrying capacity is being reached. The major current trend is for the mean size of large crocodiles to increase further, even though abundance itself is largely stable.

Management programs have had to keep pace with the changing densities of crocodiles in the wild and the changing priorities of the management agencies of the day. However, it is clear that crocodiles in the NT are now considered an important economic asset to the NT, and because of this, are secure for the future.

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**Detecting and Responding to Change in Numbers : the Future of Monitoring *Crocodylus porosus* in the Northern Territory of Australia**

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**Abstract**

All sustainable use of wildlife programs in the Northern Territory of Australia are based on three basic requirements: 1. That use of wildlife needs to be sustainable, 2. That harvested wildlife provides landholders with a commercial incentive to, 3. Undertake sound habitat management on private lands. In the case of *Crocodylus porosus*, ranching has provided landholders with an incentive to manage habitat, and resulted in recovery of the population to approximate that likely to have been present in pre-European times in northern Australia. Now we find that crocodile skin is not as valuable as it was, that tourism use of crocodiles is of major commercial value but with no direct benefit to landholders, and that safari hunting seems likely to be the best option for continuing to provide an incentive for landholders. These changes, and the fact that the Territory is now managing a population that may be close to carrying capacity, have of necessity led to review and re-evaluation of past practices in monitoring, and development of a modified program for the future. The changes and the reasons for their development are discussed in relation to the new emerging needs, and the need to maintain ongoing public accountability.

## Introduction

The first formal program for the sustainable use of wildlife in the Northern Territory was for the management of crocodiles and came into being in 1986 (Anon. 1986). Its development followed CITES endorsement of the Australian proposal for the listing of *Crocodylus porosus* in Annex II for ranching purposes (Webb *et al.*, 1984).

That management program, and all subsequent programs for the sustainable use of wildlife in the Northern Territory, was based on a set of principles that in the view of the Northern Territory Government need to be adhered to if the goals of sustainable use are to be achieved. These principles can be condensed to three major factors. They are:

1. use of wildlife must be sustainable,
2. harvests of wildlife must provide landholders with a commercial incentive to,
3. undertake sound habitat management on private lands.

The crocodile management program has adhered to these principles and experienced the desired outcome, which is the enhanced conservation of wildlife in the Northern Territory.

The incentive for landholders has been financial return from the harvest of eggs from their land. Eggs are incubated, and hatchlings raised in captivity until they reach a marketable size. Landholders participating in the program have been tolerant of large numbers of crocodiles on their wetlands, often to the point of sustaining significant losses of stock. The public has accepted the program as having good conservation outcomes backed up by an extensive program of population monitoring.

Adherence to the principles resulted in rapid growth of the Territory's population of *C. porosus* to the point where they have recovered to approximate the population present in northern Australia prior to the arrival of European man (Webb *et al.*, 1999). However successful the program may have been, the circumstances underlying that success have changed.

The relative importance of commercial values of crocodiles to the Northern Territory economy has changed. Crocodile skins are no longer as valuable as they were, and therefore the harvest of eggs no longer provide as useful a commercial incentive for landholders. There is a major commercial value in crocodiles because of tourism, but this is not as direct a benefit to landholders as the return from egg harvests. There is potential for landholders to realise commercial value from the harvest of large crocodiles and sale of skin. This value however is only a portion of the commercial incentive they could receive if safari hunting were allowed. The Northern Territory Government favours this option, but it has yet to receive the endorsement of the Australian Management Authority for CITES. This is an apparent response to the animal rights lobby.

The increased emphasis on tourism and harvest of wild adults inevitably places a greater focus on the dynamics of crocodile populations in single rivers or river systems, rather than the past emphasis on monitoring the broadly based recovery of a meta-population over many catchments across a vast area of land.

The other change results from the past success of the program. The Northern Territory is dealing with a population that is no longer rapidly growing. The population has become relatively stable, and may be close to carrying capacity. There are large numbers of crocodiles throughout the species range in the Northern Territory.

The Northern Territory's crocodile monitoring program has been reviewed because of the shift in focus of the relative commercial values of crocodiles and because of the change in population itself. The review is based upon the need to determine whether the current program is capable of detecting a relatively small change in the population of a single river within a time period commensurate with the need to, where necessary, alter management. It also examines the sensitivity of the monitoring of the meta-population. It also seeks to ensure that the effort expended in monitoring is commensurate with both the outcomes achieved, and the outcomes needed i.e. that the effort is efficient and effective.

Any such review requires that there be an *a priori* decision as to what constitutes a "relatively small change", and what constitutes a "a time period commensurate with the need, where necessary, to alter management". The small level of change in the population should be one that has historically been shown to allow the population to rapidly recover should the cause of the decline be recognised and appropriate management actions instituted.



Historically the Territory's crocodile population exhibited an annual growth of approximately 10% between 1980 and 1990, after which the rate of growth decreased considerably (Webb *et al.*, 1999). A 10% per annum change in monitored numbers was regarded as being relatively small, potentially detectable, and not so large as to require a long period for recovery should management action be required. It also corresponds to the maximum level of harvest permitted for the removal of non-hatchling crocodiles.

It is neither practical nor physically possible to detect a relatively small change (e.g. 10% per annum in a wild population) the instant it occurs, irrespective of whether the change is abrupt or gradual. The time period to be allowed for detection should have some relationship to the biology of the species being monitored. The time required to detect a small change in a population of a species of mouse should obviously be less than that to be used to detect a similar proportional change in a population of elephants. The time to be allowed to detect a small change should be related to the life history of the animal concerned, rather than designed to meet some industry, or bureaucratic imperative.

The relevant life history parameter is the animal's generation time, defined as the mean time period between birth of parents and the birth of their offspring. Unfortunately there are no reliable life or fertility tables for *C. porosus*. An approximation is Webb *et al's.*, (1987) estimate of 12 years as the average age of sexual maturity of female, wild *C. porosus*.

The *a priori* value of maximum time to detect a 10% per annum change in the monitored number was chosen as half a generation time i.e. approximately 6 years. This detection time would allow managers to implement an adjusted management regimen well prior to sexual maturity of animals born at the beginning of the period. An equivalent time for an endangered species of rat would be something on the order of 4.5 months (Bonner 1965).

## Methods

Crocodile monitoring in the NT is based on two procedures: spotlight counts in rivers and inlets, and helicopter based counts over sections of rivers. The former was first developed by Messel *et al.*, (1981) and has been religiously maintained ever since 1984 for all rivers. Some rivers have data beginning from 1975 or 1976 because of the early work of H. Messel. The second method is based on work conducted by Bayliss *et al.*, (1986) and was implemented in 1989, and maintained ever since. Data analysed included those gathered in 1999, but excluded data from some rivers taken under less than optimal conditions in 1998 .

Analyses of the data were contracted to CSIRO, Mathematical and Information Sciences.

Analyses were performed to answer the following questions:

- How long would it take to detect a decline of 10% per year in the crocodile populations of the different river systems monitored (current practice of an annual spotlight count)?
- With pooled data across all mainstream systems, how long would it take to detect a 10% decline in the population occurring in several rivers?
- How long would it take to detect the decline if counts were conducted every two years?
- How long would it take to detect the decline if counts were conducted every three years?
- In rivers where spotlight and helicopter monitoring are carried out, do the data collected by helicopter monitoring reflect trends detected by spotlight monitoring?
- How long would it take for helicopter monitoring to detect a decline in the population of 10% per year?

Data were analysed using the log of the ratio of the number of *C. porosus*/km observed by spotlight or from a helicopter. Use of the log prevented noticeable violation of the assumptions of normality, equality of variance through a sequence and absence of auto-correlation.

Desired probabilities of two types of error were specified. Type 1, ($\alpha$) the probability of identifying a change when there is none, and Type 2, ($1-\beta$) the probability of not identifying a change when there is one. The value chosen for $\alpha$ was 0.05.

Trends were detected using the assumption that there was a nominal level for the mean log-ratio of crocodiles (as observed by standard spotlighting protocol) and that the aim is to detect a drop from this level. A linear trend could have been used as the indicator but would not perform well if the actual change was some pattern other than a steady trend e.g. a sudden drop to a lower steady mean.

The method requires knowledge of the true standard deviation of the sampling error. This is taken as 'known' from the previous data from which an estimate, the residual standard deviation $s$, is obtained. There is clearly the assumption that the standard deviation for the current data is the same as estimated in the past. If this were not correct, a solution would be to insert the upper percentage point of the t-distribution in place of $z_\alpha$, and then $s$ could be estimated from the current data.

Let $n$ be the number of observations in the sequence, $d$ the log-decline rate (for 10% per annum, $d$=0.1054) and $k$ the interval between successive observations. $z_\alpha$ denotes the upper 100$\alpha$% point for the Normal distribution; for $\alpha$=0.05, $z_\alpha$=1.645. The power is given by the formula:

$$1 - \beta = \Phi\,[z_\alpha + (kd/s)\sqrt{c}],$$

where $\Phi$ is the standard Normal distribution function and $c$ is given by $c = n(n-1)/2$. The value of $z_\alpha$ has been derived from the supposition that a one-sided test is to be performed; this is appropriate because the intention is to detect a one-sided change, a decrease, not increase.



The ability of helicopter observations to predict the spotlight observations was assessed by regression of the spotlight log-ratio on the helicopter log-ratio. The statistical significance is the same as for analysing their correlation. We compared the significance of the trends in log-ratio obtained from the two monitoring techniques.

Where mainstream data were combined, the years used were those where there were data for all rivers. Otherwise the pooled mean log-ratio would be distorted according to the typical abundance in the rivers that happened to be included. This restriction resulted in a relatively small sample for analysis.

## Results

### Capacity to Detect a 10% per annum change

Tables 1, 2 and 3 provide the probability of detection (power) of a decline of 10% per annum if a hypothesis was preformed at the 5% level of significance.

Tables 1, 2 and 3 can be used to estimate the number of observations required to detect the specified trend (10%/annum) with specified power. Table 1 provides the information for annual counts, Table 2 for counts each second year and Table 3 for counts each three years. The Tables can be used as follows. If a sequence of observations has a residual standard deviation $s$=0.20, and the concern is a decline in the nominated mean log-ratio, then Table 1 shows that n=5 annual observations would have a probability of 0.7615 of producing a significant correlation from a 5% significance test. If a power of $\beta$ > 0.90 were required, then $n$=6 with a power of 0.9431 would suffice. Similarly, for a power of at least 90%, observations every 2 years, $n$=4 (i.e. years 1, 3, 5) would be required (Table 2). For observation every 3 years, $n$ = 4 (years 1, 4, 7, 10) would be needed (Table 3).

### Analysis of Mainstream Data

Table 4 gives river names and the corresponding residual standard deviation ($s$), the linear trend in of the log-ratio on time, its P value (two-sided probability), the auto-regression coefficient, and the regression slope of spotlight log-ratio on helicopter log-ratio and its one sided (upper tail) P-value.

There were clear increases in the number of crocodiles in the Daly, Liverpool and Mary rivers. Autocorrelation was never significant and not of consistent sign, and can be ignored.

The helicopter counts showed no significant association with the spotlight counts, except for the Cadell and the pooled data where the relationship was not very strong. Some of the non-significant estimates actually had a negative value. The conclusion is that overall the helicopter counts bear no useful relationship to the spotlight data.

Except for the pooled counts, Table 5 indicates that the values of $s$ from helicopter counts are beyond the range of $s$ for the spotlight data given above, and that an expected decline of 10% per annum would take a very long time to detect.

**Table 5:** Summary of the analyses for helicopter log-ratio ($s$=residual standard deviation).

| Location | $s$ | Trend (s.e.) | P-value |
|---|---|---|---|
| Adelaide downstream | 0.282 | 0.0064 (0.0268) | 0.817 |
| Adelaide upstream | 0.333 | -0.0283 (0.0402) | 0.504 |
| Blyth | 0.419 | 0.0023 (0.0400) | 0.955 |
| Cadell | 0.429 | -0.0515 (0.0409) | 0.240 |
| Daly | 0.398 | 0.0085 (0.0380) | 0.827 |
| Liverpool | 0.541 | -0.0778 (0.0515) | 0.165 |
| Mary downstream | 0.337 | 0.0221 (0.0321) | 0.509 |
| Tomkinson | 0.437 | -0.0295 (0.0417) | 0.497 |
| Pooled | 0.223 | -0.0245 (0.0269) | 0.392 |

The helicopter counts entirely failed to pick up the highly significant upward trends in the Daly, Liverpool and Mary downstream. Of these three, the Liverpool was estimated by helicopter counts as declining, though not significantly so. The fact that the helicopter sequences were shorter than the spotlight sequences is partly to blame for these differences, particularly for the Liverpool River. The 1978 data point for the Daly River had an influential effect in making the spotlight trend significant. However even since 1989, this river had a steady increase in spotlight count not detected from the helicopter data.

**Table 6:** Analysis of spotlight log-ratio for 1989-1999 only ($s$=residual standard deviation).

| Location | $s$ | Trend (s.e.) | P-value |
|---|---|---|---|
| Adelaide downstream | 0.151 | 0.0015 (0.0158) | 0.929 |
| Adelaide upstream | 0.150 | -0.0306 (0.0156) | 0.085 |
| Blyth | 0.188 | 0.0000 (0.0196) | 1.000 |
| Cadell | 0.284 | -0.0214 (0.0296) | 0.491 |
| Daly | 0.093 | 0.0435 (0.0120) | 0.009 |
| Liverpool | 0.195 | -0.0007 (0.0186) | 0.970 |
| Mary downstream | 0.070 | 0.1509 (0.0132) | <0.001 |
| Mary upstream | 0.197 | 0.0332 (0.0373) | 0.414 |
| Tomkinson | 0.180 | 0.0086 (0.0188) | 0.659 |
| Pooled | 0.087 | -0.0273 (0.0164) | 0.157 |

A better comparison of spotlighting versus helicopter counting is provided by comparison of Table 5 with Table 6, which restricts the analysis of the spotlight data to the same years as the helicopter data i.e 1989 to 1999. The upward trend in the spotlighting log-ratio for the Daly and Mary downstream are highly significant. As would be expected, the estimates of $s$ differ from those of the longer sequence of spotlighting data.



Sidestream Data

The spotlight counts for sidestreams were generally lower than they were in mainstreams, and were sometimes zero. To avoid taking the logs of zeros, these counts were regarded as being one half. This does not seriously distort the conclusion that $s$ is large (or equivalently, the counts uninformative) such that detection of any trend can not be expected to be achieved within a reasonable time (Table 7). With the exception of the Adelaide River sidestreams, there can be no confidence in detecting a decline in less than 12 years after the start of the sequence.

**Table 7:** Summary analyses of the sidestream spotlight log-ratios ($s$=residual standard deviation, AR=autocorrelation).

| Sidestream | $s$ | Trend (s.e.) | P-value | AR (s.e.) |
|---|---|---|---|---|
| Adelaide | 0.233 | 0.0200 (0.0082) | 0.026 | 0.002 (0.071) |
| Blyth | 0.327 | -0.0173 (0.0094) | 0.080 | 0.010 (0.090) |
| Atlas | 0.513 | -0.0401 (0.0161) | 0.023 | -0.013 (0.148) |
| Gudjerama | 0.396 | 0.0216 (0.0133) | 0.122 | 0.019 (0.131) |
| Maragul | 0.345 | 0.0411 (0.0108) | 0.001 | 0.027 (0.117) |
| Morngarrie | 0.537 | -0.0226 (0.0167) | 0.191 | -0.007 (0.138) |
| Mungard | 0.432 | 0.0187 (0.0135) | 0.180 | 0.020 (0.122) |
| Toms | 0.861 | 0.0725 (0.0309) | 0.034 | 0.038 (0.262) |

**Discussion**

The inescapable conclusion is that, not withstanding its considerable cost efficiencies in undertaking counts in remote areas (Bayliss *et al.*, (1986), helicopter counts are not able to provide data suitable for detecting significant changes in the Northern Territory's populations of *C. porosus* within the time constraints required by management. It is not suitable for monitoring the effects of management in individual rivers and is not as sensitive to change as spotlight count data pooled across rivers.

The commercial value of the different rivers varies according to the nature of the uses to which the crocodiles are put. In some rivers the tourism value vastly exceeds that of the egg harvest, some have harvests of adult crocodiles while others do not, and most are subject to eggs harvests. The sustainability of these uses depends upon monitoring systems sensitive to change within those rivers, as well as sensitivity to change in the meta-population distributed across river systems. The Parks and Wildlife Commission of the Northern Territory will abandon helicopter counts for the purpose of monitoring change in numbers of *C. porosus*. It may prove to be a useful adjunct to spotlight surveys in determining the nature of a change in numbers detected using spotlight counts. Helicopter counts appear to detect change in the size distribution of large crocodiles better than does spotlighting.

In general spotlight counts of side streams do not provide useful data, and will only be maintained in situations where they are cost effective, and may provide some useful information e.g. the Adelaide River.

Spotlight counts in mainstreams provide the best option for detecting change in number of *C. porosus* within acceptable periods of time.

The Parks and Wildlife Commission of the Northern Territory will continue to use spotlighting to monitor mainstream populations of *C. porosus*. Counts will continue to be conducted each year, and some rivers previously monitored from helicopter will be subject to spotlight surveys. Tables 1, 2 and 3 clearly indicate that with the existing residual standard deviations of the log-ratios, time to detection of a significant change would in general be unacceptable if counts were conducted each two or three years. These frequencies of monitoring result in detection times over, and in some cases well over, half the approximated generation time for *C. porosus*.

Table 8 combines data from Tables 1 and 4 for annual counts and a probability of detection of 0.90, with information on the extent of depletion of the population at the time of detection, and estimates of the time required for the populations to return to the pre-existing mean log-ratio. Spotlighting provides a sensitive method of detecting an overall change in the number of the *C. porosus* meta-population. A change of 10% can be detected in 4 years, with a concomitant 33% reduction in the population and period of 9 years required for full recovery following removal of the factor causing decline. This is detection within one third, and recovery within two thirds of a generation time. The full recovery of the population from its very low point at the cessation of uncontrolled shooting took approximately 19 years. This is equivalent to 1.6 generation times, which is remarkably fast. It would be interesting to know whether any monitoring program for an endangered species had this level of sensitivity at the meta-population level. The equivalent times for detection of change and subsequent recovery for an endangered species of rat would be 3 and 6 months respectively.

**Table 8:** The time to detect a 10% per annum change, extent of the population declines at detection and the recovery period in mainstream populations, and the pooled populations, using an annual spotlight survey with $\beta = 0.90$.

| River | *s* | Detection Time (years) | Decline (%) | Recovery Time (years) |
|---|---|---|---|---|
| Adelaide (down) | 0.17 | 6 | 47 | 11 |
| Adelaide (up) | 0.21 | 6 | 47 | 11 |
| Blyth | 0.20 | 6 | 47 | 11 |
| Cadell | 0.23 | 7 | 52 | 12 |
| Daly | 0.12 | 5 | 41 | 9 |
| Liverpool | 0.16 | 5 | 41 | 9 |
| Mary (down) | 0.20 | 6 | 47 | 11 |
| Mary (up) | 0.22 | 6 | 52 | 11 |
| Tomkinson | 0.24 | 7 | 48 | 12 |
| Pooled | 0.11 | 4 | 33 | 8 |

As anticipated the sensitivity of detection of change in individual mainstreams is less than that for the meta-population, and is strongly influenced by the size of the residual standard deviation. None the less, only two of nine mainstreams have a detection time



greater than half a generation time. Both these rivers had detection times of 0.58 of a generation (not markedly greater than 0.5). All mainstreams would recover within a single generation time.

While the times to detection of change and the recovery periods are biologically acceptable, any shortening in these time periods would clearly be advantageous. This could only be achieved by gaining a better understanding of the causes of variance in the spotlight counts and so be able to reduce the residual standard deviation. This will be investigated over the next few years. As well as benefiting industry, a significant reduction is *s* would allow a reduction in the frequency of spotlight surveys without loss of sensitivity.

These analyses provide an understanding of the capacity of the monitoring methods to detect change, they do not provide an effective decision making tool. Crocodile monitoring is equivalent to quality control of an industrial process using a sampling scheme. Industry uses decision rules for action when a process is believed to have slipped or drifted out of control. There is a large array of methods available (Bowker and Lieberman 1959; Davies and Goldsmith 1972; Montgomery 1991). The Parks and Wildlife Commission of the Northern Territory has initiated development of such a tool.

## Conclusions

Harvest levels of *C. porosus* in the Northern Territory of Australia are inherently conservative. Introduction and continuation of harvests and tourism use of the meta-population was accompanied by a dramatic recovery of the meta-population. The population is currently relatively stable with continuing growth in some populations. While a significant decline in either a single population or the meta-population seems very unlikely, it is critical that monitoring be designed to meet the needs of public accountability, as well as to deal with the remote probability of there being some form of decline. This means focusing on demonstrating that existing management practice is sensitive to change in populations subject to use, and sensitive to changes in circumstance both biological and sociological.

It is critical that monitoring, and the parameters developed for decision making be grounded in biological characteristics relevant to the species concerned, not industrial or bureaucratic dictates. There is a need for sound statistically based monitoring and decision systems. Annual spotlighting counts currently meet these needs. Efforts will be made to improve the sensitivity of monitoring over the next few years, and a sound decision making tool is being developed.

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**Table 1:** Probability ($\beta$) of detection of a 10% per annum change from a nominated mean log-ratio of a population in relation to the residual standard deviation ($s$) and number (n) of samples using annual spotlighting counts.

| n | | | Power | | | | |
|---|---|---|---|---|---|---|---|
| *s* | 3 | 4 | 5 | 6 | 7 | 8 | 9 |
| 0.10 | 0.5714 | 0.9352 | 0.9989 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.11 | 0.5056 | 0.8904 | 0.9958 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.12 | 0.4506 | 0.8387 | 0.9887 | 0.9999 | 1.0000 | 1.0000 | 1.0000 |
| 0.13 | 0.4047 | 0.7842 | 0.9761 | 0.9995 | 1.0000 | 1.0000 | 1.0000 |
| 0.14 | 0.3664 | 0.7300 | 0.9574 | 0.9985 | 1.0000 | 1.0000 | 1.0000 |
| 0.15 | 0.3342 | 0.6781 | 0.9327 | 0.9961 | 1.0000 | 1.0000 | 1.0000 |
| 0.16 | 0.3070 | 0.6295 | 0.9032 | 0.9915 | 0.9998 | 1.0000 | 1.0000 |
| 0.17 | 0.2839 | 0.5849 | 0.8701 | 0.9842 | 0.9995 | 1.0000 | 1.0000 |
| 0.18 | 0.2640 | 0.5443 | 0.8347 | 0.9738 | 0.9987 | 1.0000 | 1.0000 |
| 0.19 | 0.2469 | 0.5075 | 0.7982 | 0.9600 | 0.9971 | 0.9999 | 1.0000 |
| 0.20 | 0.2320 | 0.4743 | 0.7615 | 0.9431 | 0.9944 | 0.9998 | 1.0000 |
| 0.21 | 0.2189 | 0.4444 | 0.7254 | 0.9233 | 0.9903 | 0.9996 | 1.0000 |
| 0.22 | 0.2074 | 0.4176 | 0.6904 | 0.9011 | 0.9845 | 0.9990 | 1.0000 |
| 0.23 | 0.1973 | 0.3934 | 0.6568 | 0.8770 | 0.9768 | 0.9981 | 0.9999 |
| 0.24 | 0.1882 | 0.3715 | 0.6249 | 0.8516 | 0.9671 | 0.9965 | 0.9999 |
| 0.25 | 0.1801 | 0.3518 | 0.5948 | 0.8253 | 0.9555 | 0.9943 | 0.9997 |

**Table 2:** Probability ($\beta$) of detection of a 10% per annum change from a nominated mean log-ratio of a population in relation to the residual standard deviation ($s$) and number (n) of samples using spotlighting counts each second year.

| n | | | Power | | | |
|---|---|---|---|---|---|---|
| *s* | 2 | 3 | 4 | 5 | 6 | 7 |
| 0.10 | 0.4385 | 0.9775 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.11 | 0.3858 | 0.9529 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.12 | 0.3434 | 0.9187 | 0.9999 | 1.0000 | 1.0000 | 1.0000 |
| 0.13 | 0.3090 | 0.8775 | 0.9994 | 1.0000 | 1.0000 | 1.0000 |
| 0.14 | 0.2808 | 0.8320 | 0.9980 | 1.0000 | 1.0000 | 1.0000 |
| 0.15 | 0.2574 | 0.7848 | 0.9949 | 1.0000 | 1.0000 | 1.0000 |
| 0.16 | 0.2377 | 0.7377 | 0.9894 | 1.0000 | 1.0000 | 1.0000 |
| 0.17 | 0.2211 | 0.6922 | 0.9809 | 1.0000 | 1.0000 | 1.0000 |
| 0.18 | 0.2069 | 0.6491 | 0.9691 | 0.9998 | 1.0000 | 1.0000 |
| 0.19 | 0.1947 | 0.6088 | 0.9537 | 0.9995 | 1.0000 | 1.0000 |
| 0.20 | 0.1841 | 0.5714 | 0.9352 | 0.9989 | 1.0000 | 1.0000 |
| 0.21 | 0.1748 | 0.5371 | 0.9139 | 0.9978 | 1.0000 | 1.0000 |
| 0.22 | 0.1666 | 0.5056 | 0.8904 | 0.9958 | 1.0000 | 1.0000 |
| 0.23 | 0.1594 | 0.4769 | 0.8651 | 0.9929 | 1.0000 | 1.0000 |
| 0.24 | 0.1529 | 0.4506 | 0.8387 | 0.9887 | 0.9999 | 1.0000 |
| 0.25 | 0.1471 | 0.4266 | 0.8116 | 0.9832 | 0.9998 | 1.0000 |

**Table 3:** Probability (β) of detection of a 10% per annum change from a nominated mean log-ratio of a population in relation to the residual standard deviation ($s$) and number (n) of samples using spotlighting counts each third year.

| | Power | | | | | |
|---|---|---|---|---|---|---|
| n | 2 | 3 | 4 | 5 | 6 | 7 |
| $s$ | | | | | | |
| 0.10 | 0.7225 | 0.9999 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.11 | 0.6506 | 0.9996 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.12 | 0.5862 | 0.9982 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.13 | 0.5297 | 0.9949 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.14 | 0.4807 | 0.9883 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.15 | 0.4385 | 0.9775 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.16 | 0.4021 | 0.9622 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.17 | 0.3707 | 0.9424 | 1.0000 | 1.0000 | 1.0000 | 1.0000 |
| 0.18 | 0.3434 | 0.9187 | 0.9999 | 1.0000 | 1.0000 | 1.0000 |
| 0.19 | 0.3197 | 0.8919 | 0.9996 | 1.0000 | 1.0000 | 1.0000 |
| 0.20 | 0.2990 | 0.8627 | 0.9990 | 1.0000 | 1.0000 | 1.0000 |
| 0.21 | 0.2808 | 0.8320 | 0.9980 | 1.0000 | 1.0000 | 1.0000 |
| 0.22 | 0.2647 | 0.8006 | 0.9962 | 1.0000 | 1.0000 | 1.0000 |
| 0.23 | 0.2504 | 0.7690 | 0.9934 | 1.0000 | 1.0000 | 1.0000 |
| 0.24 | 0.2377 | 0.7377 | 0.9894 | 1.0000 | 1.0000 | 1.0000 |
| 0.25 | 0.2264 | 0.7071 | 0.9841 | 1.0000 | 1.0000 | 1.0000 |



**Table 4:** Summary of the analysis of spotlighting log-ratio ($s$=residual standard deviation, AR=autocorrelation).

| Location | $s$ | Trend (s.e.) | P-value | AR (s.e.) | Helicopter (s.e.) | P-value |
|---|---|---|---|---|---|---|
| Adelaide downstream | 0.167 | 0.0149 (0.0097) | 0.148 | 0.0006 (0.0653) | 0.075 (0.182) | 0.346 |
| Adelaide upstream | 0.213 | 0.0273 (0.0081) | 0.004 | 0.0132 (0.0492) | 0.182 (0.215) | 0.215 |
| Blyth | 0.195 | 0.0183 (0.0057) | 0.004 | 0.0076 (0.0570) | -0.203 (0.136) | 0.923 |
| Cadell | 0.225 | -0.0017 (0.0066) | 0.804 | -0.0014 (0.0614) | 0.407 (0.176) | 0.025 |
| Daly | 0.116 | 0.0680 (0.0060) | <0.001 | 0.0386 (0.0255) | 0.072 (0.154) | 0.327 |
| Liverpool | 0.163 | 0.0262 (0.0048) | <0.001 | 0.0191 (0.0478) | -0.051 (0.106) | 0.679 |
| Mary downstream | 0.202 | 0.1025 (0.0122) | <0.001 | 0.0587 (0.0757) | -0.027 (0.504) | 0.520 |
| Mary upstream | 0.218 | 0.1224 (0.0131) | <0.001 | 0.0502 (0.0621) | -- | -- |
| Tomkinson | 0.237 | 0.0177 (0.0074) | 0.027 | 0.0102 (0.0532) | -0.201 (0.116) | 0.939 |
| Pooled | 0.107 | 0.0171 (0.0071) | 0.038 | 0.0047 (0.0345) | 0.263 (0.112) | 0.040 |

# Effects of Egg and Hatchling Harvest on American Alligators in Florida

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*Abstract:* Harvest of crocodilian eggs and young for captive rearing (ranching) has been used worldwide as an option for producing crocodilian skins and meat from wild stock. The long-term effects of harvesting a certain proportion of early age class, wild American alligators (*Alligator mississippiensis*) without repatriation is unknown. We removed an estimated 50% of annual production of alligators on Lakes Griffin and Jesup in central Florida over an 11-year period and monitored population levels via night-light counts. Densities of the total alligator population increased ($P < 0.037$) on all areas. Count densities of adult (≥183 cm total length [TL]) alligators increased ($P < 0.003$) on harvest areas but remained stable ($P = 0.830$) on the control (no harvest) area, Lake Woodruff National Wildlife Refuge (Lake Woodruff NWR). Observed densities of juvenile (<122 cm TL) alligators remained stable ($P > 0.117$), and subadult (122--182 cm TL) alligators increased ($P < 0.011$) on harvest areas. The density of juveniles on the control area increased ($P = 0.006$), and the density of subadults showed some evidence of increasing ($P = 0.088$). No changes were detected in size distributions on the treatment areas. Nest production, as observed from aerial helicopter surveys, increased ($P < 0.039$) on Lake Woodruff NWR and Lake Jesup and showed some evidence of an increase on Lake Griffin ($P = 0.098$) during 1983--91. A 50% harvest rate of eggs or hatchlings did not adversely affect recruitment into the subadult or adult size classes.

For complete details of the study, see:

Rice, K. G., H. F. Percival, A. R. Woodward, and M. L. Jennings. 1999. Effects of egg and hatchling harvest on American alligators in Florida. J. Wildl. Manage. 63:1193-1200.

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**INFORME DE INVESTIGACION**

## Análisis de la explotación del Caimán común o Babilla (*Caiman crocodilus*) en la Isla de la Juventud, Cuba.

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### 1. Introducción

Las especies de reptiles pertenecientes al grupo de los cocodrilos (cocodrilos, caimanes y gaviales) han sido tradicionalmente explotadas por el hombre como un valioso recurso natural, principalmente por su piel y carne. La situación actual de la explotación comercial de estos grandes reptiles, con fines de conservación y uso sostenible, ha sido revisada por Thorbjarnarson (1999), quien hace un recuento histórico de las altas y [redacted] bajas de dicha explotación y concluye que el problema clave está en que un programa de conservación no puede estar basado exclusivamente en la venta de un solo producto (piel) para un mercado de lujo y **aconseja** la diversificación del mismo con la venta de otros productos y el ecoturismo, así como prestar un mayor apoyo a las especies amenazadas de extinción.

Entre las especies de cocodrilos y caimanes explotadas con éxito, se encuentra el caimán común, caimán de anteojos, baba o babilla (*Caiman crocodilus*) en Venezuela y otros países sudamericanos. La explotación de esta especie es básicamente por su piel, considerada sin embargo de inferior calidad que la del cocodrilo; no obstante puede generar beneficios económicos significativos, ya que entre 1983 y 1995 en Venezuela, más de un millón de babas fueron cosechadas con un valor de exportación de más de 115 millones de dólares. Sobre dicha explotación en este país existe una abundante literatura, resumida por Thorbjarnarson y Velasco (1998, 1999), quienes consideran que la mayoría de los indicadores sugieren que las cosechas han sido sostenibles en relación a las poblaciones de babas. Para los propietarios de tierra con esta especie, su explotación produce un alto retorno a la inversión, pero con ganancias inferiores a las del ganado. La explotación de esta especie no genera incentivos para la protección de su hábitat, ya que este es el mismo de la ganadería, pero sí ha generado fondos para las agencias gubernamentales que manejan la vida silvestre de Venezuela.

En Cuba en la actualidad existen tres especies del grupo de los cocodrilos, el cocodrilo cubano (*Crocodylus rhombifer*), endémico que solo vive en la Ciénaga de Zapata y fue extirpado de la Ciénaga de Lanier, pero hoy ha sido de nuevo reintroducido aquí (Rodríguez, 1996; Ross, 1997); el cocodrilo americano (*C. acutus*) ampliamente distribuido por toda Cuba y el Caribe y la baba, babilla o caimán común, introducido en la Isla de la Juventud en 1959.

El caimán o babilla se encuentra hoy ampliamente distribuido y es abundante por toda la Isla de la Juventud, sobre todo en presas y micropresas de la parte

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El caimán o babilla se encuentra hoy ampliamente distribuido y es abundante por toda la Isla de la Juventud, sobre todo en presas y micropresas de la parte norte. Se ha estimado una población superior a los 40 000 individuos (Rodríguez, 1996). Aunque al inicio se pensó que esta especie pudo haber causado la extirpación de *C. rhombifer* de la ciénaga de Lanier, hoy se sabe que esta especie desapareció antes de que los caimanes se hicieran abundantes en dicha ciénaga, en donde por otra parte no han sido numerosos (Rodríguez, 1996; Ross, 1997). La ecología de esta especie se encuentra resumida en Thorbjarnarson (1991).

Desde 1995 se lleva a cabo un programa de explotación de esta especie de caimán introducido, en parte para obtener beneficios económicos y en parte para controlar las cantidades de caimanes. Los ingresos económicos de este programa fueron utilizados para apoyar un incremento en la caza de caimanes, así como acciones para la conservación de los cocodrilos cubanos (Rodríguez, 1996). El objetivo de nuestra investigación fue valorar los resultados obtenidos de dicha explotación en un período de cuatro años (1995-1998) en cuanto a tres aspectos de su uso sostenible, siguiendo a Prescott-Allen y Prescott-Allen (1996):

- Sostenibilidad demográfica. ¿Se han mantenido estables las poblaciones de babillas en cuanto a su tamaño y otros aspectos demográficos o han sufrido depletación y alteraciones?
- Sostenibilidad ecológica. ¿Se afecta o no el hábitat de las babillas con su explotación?
- Sostenibilidad económica. ¿Es realmente económica la explotación de la babilla, en términos de relación costos/beneficios?

Un cuarto aspecto de la sostenibilidad, la social, que se relaciona con los beneficios que aporta a las comunidades la explotación de las babillas, será tratado en otro informe.

Nuestra investigación se apoya en dos trabajos previos sobre la especie, uno referido a la estima de su población total previa a la explotación (Méndez et al, 1994) y otro sobre la valoración económica de dicha explotación (Savón, 1998). Los resultados obtenidos se comparan con los registrados para la explotación de la misma especie en Venezuela (Thorbjarnarson y Velasco, 1998, 1999).

## 2. Materiales y métodos

Durante los cuatro años de explotación de la babilla en la Isla de la Juventud, se extrajeron animales de varias localidades de la parte norte de la isla y de la ciénaga de Lanier. Muchas de estas localidades fueron comunes en años consecutivos, así en 1996, seis localidades (20.7%) también fueron explotadas en 1995; en 1997, lo fueron ocho (30.7%) con respecto a 1996 y en 1998 solo dos localidades (18.2%) en relación a 1997.

Los hábitat de capturas en esas localidades fueron ciénagas de inundación periódica y lagunas naturales en la ciénaga de Lanier y presas y micropresas en la parte norte, hábitat decrito por Méndez et al (1994).

Según el estimado poblacional de 1993 (Mendez et al, 1994) existían en la Isla de la Juventud (parte norte y ciénaga de Lanier) unos 25 500 caimanes sobre la base de una densidad de 0.57 a 15 caimanes/ha según hábitat, de los cuales 6 746 eran grandes (más de 150 cm de longitud total), 10 170 medianos (de 90 a 150 cm de longitud total) y 8 585 pequeños (menos de 90 cm de longitud



total). Basándose en estos datos se planificó una extracción inicial anual (1995-96) de 2 000 animales mayores de 90 cm de largo total, es decir los medianos y grandes (total: 16 916). Esta cifra de 2 000 individuos por año representa el 7.8% de la población total y el 11.8% de la población a explotar, cifras por debajo del potencial máximo calculado para la especie en la isla que es del 22.5% (Méndez et al, 1994). Para los años 1997-98, la cifra de extracción se bajó a 500 animales/año, dada las limitaciones de recursos materiales para las capturas.

**Tabla 1. Comparación de las metodologías de explotación del caimán en Venezuela y Cuba.**

| Aspectos | Venezuela | Cuba |
|---|---|---|
| Producto primario | Piel procesada | Piel salada |
| Producto secundario | Carne | Carne |
| Lugares de captura | Tierras privadas | Tierras estatales |
| Hábitat de captura | Antrópicos (zonas ganaderas) | Antrópicos y naturales |
| Epoca de capturas y horarios | Enero a Abril - nocturno | Casi todo el año - nocturno |
| Talla mínima sexo mayormente capturado | 180 cm - machos | 90 cm - machos |
| Modo de captura | Arpón | Lazo, arpón, fusil |
| Destino de los productos: Piel<br>Carne | Comercio internacional<br>Comercio local | Comercio internacional<br>Comercio local |
| Tasa de extracción (%)<br>Pob. Total | 7 | 7.8 (1995-96)<br>1.9 (1997-98) |
| Pob. a explotar | 20 | 11.8 (1995-96)<br>2.9 (1997-98) |
| Densidades (caimanes/ha) | 0.09 - 0.39 | 0.57 - 15 |
| Caza ilegal | Sí | Sí |
| Uso sostenible | Sí | ? |
| Monitoreo previo a las capturas | Sí | No |

En la tabla 1 se presenta un resumen de los aspectos más importantes de la metodología empleada en las capturas de babillas, así como su comparación con la empleada en Venezuela en los últimos años para la misma especie. Siete aspectos fueron iguales en ambos casos, los que se refieren al producto primario (piel) y secundario (carne) por el cual se explota la especie, al destino de estos productos, al horario de captura, a la tasa de extracción en relación a la población total (ligeramente más alta al inicio en nuestro caso) a la mayor captura de machos y al problema de la caza ilegal, en cuanto por supuesto a su existencia, pero no a su magnitud. Para nuestro caso esta caza ilegal se estima en una extracción anual de unos 400-500 animales por año, pero aquí se incluyen también el cocodrilo americano, por lo que por el momento no podemos estimar su impacto específicamente sobre las poblaciones de babillas.

Ocho aspectos de nuestra metodología resultaron diferentes con respecto a la empleada en Venezuela, estos fueron los siguientes: Los lugares de capturas en nuestro caso se hicieron solamente en tierras estatales, los hábitat de captura fueron tanto sitios antropizados (presas y micropresas) como naturales (ciénaga) y nuestra época de captura se realizó durante todo el año. O sea, con respecto a los dos últimos aspectos, fuímos más "amplios" en comparación con la metodología venezolana, lo que se comprende pues nuestra población total es mucho menor. Por esta misma razón, nuestra talla mínima de captura fue muy inferior a la venezolana. Aunque al principio se utilizaron arpones para las capturas de los animales como en Venezuela, después se cambió para el fusil. Nuestra tasa de extracción con respecto a la población total disminuyó en los dos últimos años y con respecto a la población a explotar fue siempre muy inferior a la planteada para Venezuela; sin embargo, nuestras estimas de densidades, sobre las que se basa la explotación, son mucho mayores. Por último, un grave defecto de nuestra metodología, que sí posee la de Venezuela, es el monitoreo previo de las poblaciones antes de las extracciones. En esto influyó el hecho de que las extracciones de babillas se hacían con el fin específico de bajar la densidad, así que en principio no importaba mucho si esta se afectaba o no con las extracciones.
La explotación de babas en Venezuela se considera sostenible (Thorbjarnarson y Velasco, 1998) y este es el aspecto objeto de estudio de esta investigación, en relación a las babas de la Isla de la Juventud.

Tabla 2. **Variables analizadas en el estudio de la explotación del caimán en la Isla de la Juventud.**

| Variables originales | Datos en años 95 | 96 | 97 | 98 | Variables en el análisis | Análisis estadísticos | Prueba para sostenibilidad |
|---|---|---|---|---|---|---|---|
| Cantidad de animales /sexo/mes | X | X | X | X | Indice de captura Cociente sexual | ANOVA bifactorial Prueba G | Demográfica |
| Longitud total (cm) /sexo/mes | X | X | X | X | Longitud total (cm) Clasesde tamaños | ANOVA bifactorial (año x sexo) Prueba G | Demográfica |
| Peso vivo/peso carcasa (kg) | - | X | - | - | Rendimiento en canal (%) | ANOVA bifactorial (sexo x estación) | Económica |
| Ancho de la piel (cm) | - | - | X | X | Ancho relativo de la piel (%) | ANOVA bifactorial (estación x año) | Económica |
| Calidad de la piel | - | - | X | - | 1ra y 2da calidad | - | Económica |

La tabla 2 resume las variables que fueron analizadas en este estudio. En total fueron seis variables pero no todas se registraron en los cuatro años de estudio. Las cantidades de animales capturados por sexo y mes están



completas para todos los años y de ellas se derivaron dos variables para el análisis estadístico, el índice de captura y el cociente sexual. Obviamente el total de animales capturados por mes depende básicamente del esfuerzo de captura, que está determinado por el número de capturadores, el tiempo empleado y el área explotada. Los capturadores actuaban en grupos de 3-4 hombres y estos siempre se mantuvieron [redacted] constantes a lo largo de todos los años, por consiguiente solo quedaban lós efectos del tiempo y el espacio, que se dieron en términos de días de captura (con aproximadamente las mismas horas totales) y número de lugares de capturas. Para poder entonces comparar las capturas por meses, se calculó el índice de captura como animales capturados /día en captura/número de localidades de capturas. Los valores mensuales de este índice de captura fueron entonces comparados entre años y estaciones del año (seca y lluvia) por análisis de varianza (ANOVA) bifactorial (3 años x 2 estaciones).
Con los datos de animales capturados por sexo, se calculó el cociente sexual para cada año, dado como porcentaje de machos, el que fue sometido a una prueba G para su análisis estadístico.
La segunda variable con datos completos en los cuatro años fue el largo total (cm) de los animales capturados, que entró en los análisis estadísticos como tal y como clases de tamaños. En el primer caso se analizaron las diferencias entre años (considerando el sexo) de los valores medios del largo total por un análisis de varianza bifactorial (4 años x 2 sexos) y en el segundo se analizaron las frecuencias de clase entre años y sexo por una prueba G triple, considerando tres clases de tamaño (110-120, 121-139 y 140 cm o más del largo total). Con estas cuatro variables se intentaba verificar la sostenibilidad demográfica de la población de babilla, en el sentido de que si esta se cumple: a) no existen cambios en el índice de captura mensual por año, o sea no hay signos de depletación; b) no hay cambios en la estructura de la población en relación al cociente sexual por año, largo total medio anual y frecuencia de clase de tamaño por años.
Las cuatro restantes variables solo fueron registradas en algunos años, por lo que brindan una información limitada sobre la sostenibilidad de la explotación de la babilla, en este caso sostenibilidad económica. Los pesos vivos del animal y de su carcasa o canal (el animal libre de piel, cabeza, extremidades y vísceras) nos permitió calcular la variable rendimiento en canal como (peso vivo/peso canal) x 100, lo que se analizó solo en el año 1996, para los efectos sexo x estación por un análisis de varianza bifactorial (2 sexos x 2 estaciones). El ancho de la piel se analizó como ancho relativo al largo total del animal para fines comparativos, calculando el cociente: (ancho de piel/largo total) x 100; los valores promedios anuales de esta variable se compararon por un análisis de varianza bifactorial (2 estaciones x 2 años) para los años 1997 y 1998, considerando el efecto estación (seca y lluvia) previa verificación del no efecto del sexo. La última variable considerada fue la calidad de la piel, solo registrada en los años 1996 y 1997 y no sometida a ningún análisis estadístico. La sostenibilidad económica se demuestra con estas variables, en el sentido de que la explotación no disminuye sus valores promedios anuales (rendimiento en canal, ancho de piel relativo) o su frecuencia (piel de primera calidad). Esto

realmente solo se pudo verificar para el ancho relativo de la piel y solo para los dos últimos años de la explotación.
Para todos los análisis de varianza se hicieron verificaciones previas de la distribución normal del carácter y de la homogeneidad de varianza entre los tratamientos. Todos los análisis estadísticos siguen a Sokal y Rolf (1981) y se realizaron con el programa STATISTICA.

## 3. Resultados y Discusión

La cantidad total de animales capturados por mes y año, separados por sexos, así como los días en captura, el número de localidades de capturas y el índice de captura por meses y años, se presentan en las tablas 3, 4, 5 y 6 (no existían registros de localidad para 1995 por lo que el índice no se pudo calcular para este año). Las capturas totales casi siempre estuvieron por debajo de lo planificado (2000 animales para 1995-96; 500 para 1997-98) excepto en 1997. Durante los tres primeros años hubo un incremento notable de animales capturados, pero estos decayeron abruptamente en 1998; esto no fue debido a las capturas anteriores, sino a la falta de recursos materiales, que disminuyeron las posibilidades de horas y localidades de captura. El índice de captura promedio por mes presentó distribución normal y fluctuó entre 0.29 y 3.33 con valores extremos de 4, 66, 7 y 9. Dicho índice comenzó bajo en 1996 (valor promedio de 1.1 caimán/localidad/día) pero después se incrementó hasta llegar a 1.9 caimanes/día/localidad (tabla 7). Sin embargo estas diferencias no fueron estadísticamente significativas por el análisis de varianza. Por contraste, los valores promedios del índice entre estaciones de seca y lluvia si lo fueron, con el mayor valor (1.83) en seca como era de esperar, ya que aquí los animales se concentran más y son más fáciles de capturar. Igualmente, la regresión del índice de captura mensual en los meses de captura (para los tres años) produjo un valor de regresión estadísticamente significativo y positivo, indicando un ligero incremento del mismo (0.043 unidades del índice/mes) con un coeficiente de determinación relativamente bajo (14.6%). Estos resultados evidencian que, independiente del número total de animales capturados, el índice de captura se mantuvo igual entre los años, con un ligero incremento, si consideramos todos los años. O sea, no hay signos de depletación de la población de babas, cuando consideramos toda el área de explotación.

**Tabla 3. Animales capturados por meses y sexo, en un estudio de la explotación del caimán, Isla de la Juventud. Año 1995. M: machos, H: hembras, T: total, D: días en captura, L: localidades de captura, I: índice de captura.**

| | MESES | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | M | A | M | J | A | S | O | N | Total |
| M | 6 | 39 | 100 | 2 | 11 | 16 | 17 | 4 | 195 |
| H | 8 | 16 | 45 | 0 | 7 | 4 | 3 | 3 | 86 |
| T | 14 | 55 | 145 | 2 | 18 | 20 | 20 | 7 | 281 |
| D | 2 | 5 | 14 | 2 | 2 | 5 | 4 | 1 | 35 |
| L | - | - | - | 2 | 3 | 3 | 3 | 1 | - |
| I | - | - | - | 0.5 | 3.0 | 1.33 | 1.66 | 7.0 | - |



**Tabla 4. Animales capturados por meses y sexo, en un estudio de la explotación del caimán, Isla de la Juventud. Año 1996. Simbología como en la tabla 3.**

| | MESES | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | F | M | A | M | J | J | A | S | O | N | Total |
| M | 45 | 32 | 31 | 58 | 106 | 39 | 29 | 16 | 37 | 40 | 433 |
| H | 10 | 8 | 8 | 15 | 30 | 4 | 7 | 4 | 0 | 15 | 101 |
| T | 55 | 40 | 39 | 73 | 136 | 43 | 36 | 20 | 37 | 55 | 534 |
| D | 5 | 6 | 8 | 7 | 23 | 16 | 8 | 6 | 8 | 8 | 95 |
| L | 5 | 6 | 7 | 9 | 15 | 9 | 8 | 5 | 2 | 4 | 29 |
| I | 2.20 | 1.11 | 0.69 | 1.15 | 0.39 | 0.29 | 0.56 | 0.66 | 2.31 | 1.71 | 0.19 |

**Tabla 7. Medias (x), desviación estándar (S) y coeficiente de variación (CV) por años y estaciones para un índice de captura del caimán, en la Isla de la Juventud.**

| | Indice de captura (caimanes/localidad/día) | | | | |
|---|---|---|---|---|---|
| Años | N | X | S | CV | F del ANOVA |
| 1996 | 10 | 1.10 | 0.73 | 66.39 | 1.86 n.s |
| 1997 | 12 | 1.36 | 0.99 | 73.23 | |
| 1998 | 9 | 1.90 | 1.24 | 65.12 | |
| Estación | | | | | |
| Lluvia | 17 | 1.11 | 0.80 | 72.52 | 4.66 (p<0.01) |
| Seca | 14 | 1.83 | 1.14 | 62.27 | |
| Regresión del índice en los meses de captura<br>N=31<br>Y=0.750 + 0.043X ($S_b$=0.002)<br>$R^2$=0.146 F=4.9 (p<0.05) | | | | | |

**Tabla 8. Extracciones de caimanes por meses de diferentes años en cinco localidades de la Isla de la Juventud.**

| | Capturas/mes | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Localidades | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | 9 | 10 | 11 | 12 |
| Cayo Redondo | 1 | 4 | 6 | 7 | 8 | 7 | 28 | 3 | 5 | 13 | 5 | 2 |
| Lag. Grande | 3 | 3 | 2 | 3 | 18 | 18 | 8 | 5 | 20 | 2 | 8 | 9 |
| Los Monos | 9 | 5 | 12 | 11 | 20 | 7 | 32 | 14 | 18 | - | - | - |
| La Pasadita | 11 | 12 | 23 | 2 | 12 | 15 | 4 | 3 | 3 | - | - | - |
| La Calzada | 14 | 18 | 16 | 16 | 7 | 1 | 1 | 3 | 3 | 1 | - | - |

Igual falta de depletación se evidencia si se analizan localidades específicas en cuanto a animales extraídos por meses (sin corregir para días de captura). En cinco de estas localidades analizadas (tabla 8) no hay signos evidentes de decline, excepto quizás las localidades de La Pasadita y La Calzada, donde hubo extracciones continuas de más de 10 animales/meses.
La explotación de las babillas tampoco parece afectar su cociente sexual, si admitimos que el de las capturas refleja el de las poblaciones en la naturaleza. Excepto en 1996, donde hubo un incremento significativo de machos en las capturas, el resto de los años el porcentaje de machos capturados (entre 66 y 69%) no difiere significativamente (tabla 9). Así casi un 70% de los animales capturados son machos, lo que asemeja nuestras capturas a las efectuadas en