Australian Government
Director of National Parks

Christmas Island Biodiversity Monitoring Program: December 2003 to April 2007

Report to the Department of Finance and Deregulation,
from the Director of National Parks
September 2008

Project Contributions

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Contents

List of tables

List of figures

Executive summary

The Christmas Island Biodiversity Monitoring Program (BMP) was established to monitor the effects on biodiversity of constructing an Immigration Reception and Processing Centre (IRPC) at North West Point on Christmas Island. Construction started in July 2004.

The program revealed an apparent decline in many island species, but the precise causes are unknown and could be due to a number of contributing factors including invasive species (ant species, feral cats, black rats and the wolf snake) and habitat loss associated with human activities. In particular, the yellow crazy ant *Anoplolepis gracilipes* has had severe impacts on fauna and flora in recent years and attempts to control it remain an ongoing issue of high priority for Parks Australia.

The main impacts from the construction of the IRPC include:

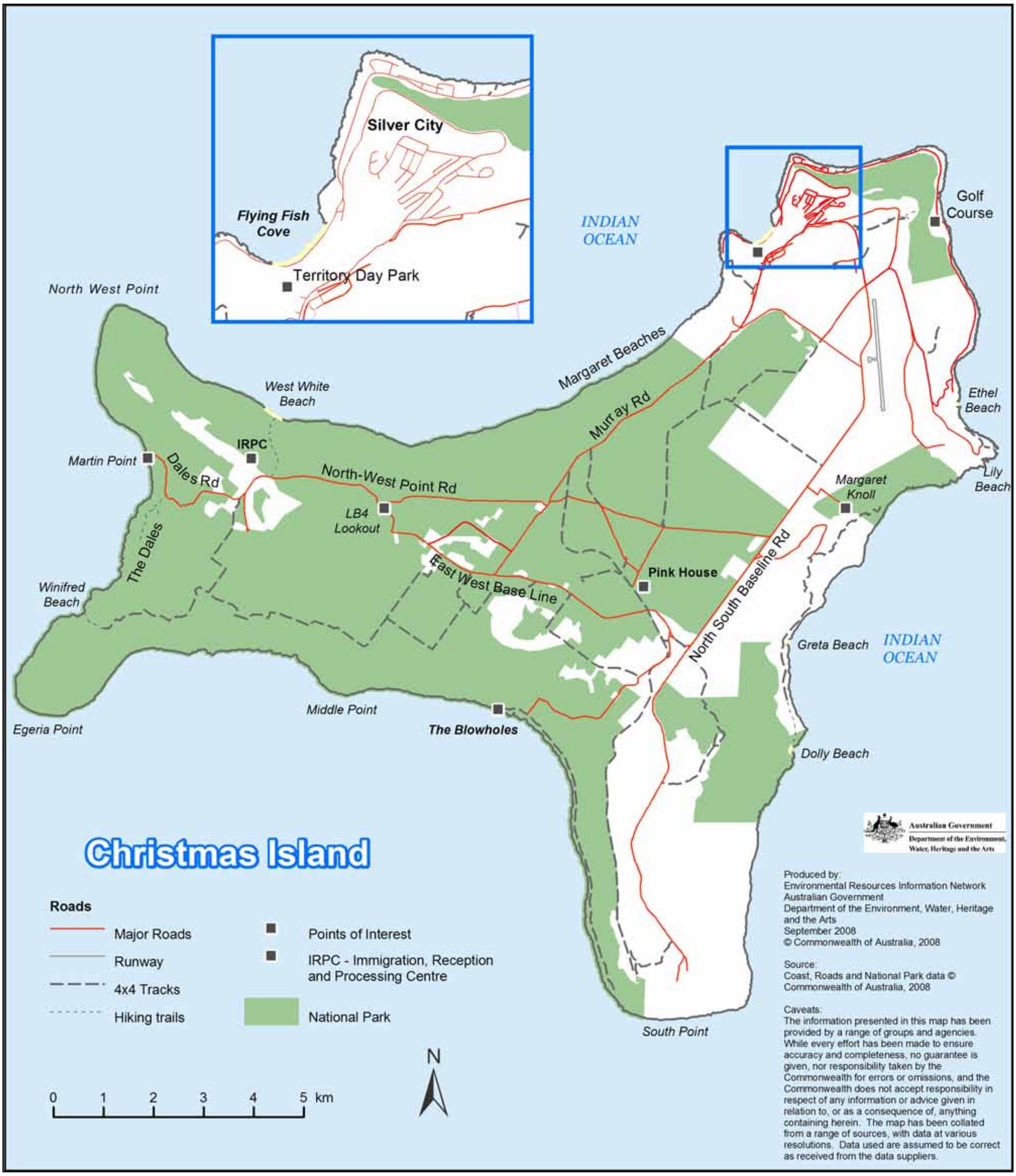
- damage to the foraging habitat of the Christmas Island pipistrelle
- increased road mortality of robber crabs and red crabs
- increased road mortality of birds
- disturbance of forest along national park boundaries.

Ongoing impacts are expected when the IRPC is operating. These are associated with lighting around the facility, affecting nesting of Abbott's boobies and pipistrelle foraging, landscaping activities bringing in weeds, an increase in the threat of invasive species as supplies are shipped in, and problems with stormwater flowing from the site into the national park. Road mortality rates are expected to remain high due to increased commuting traffic.

Recommendations to mitigate the environmental impacts of the IRPC are:

- a long-term program should be established to monitor the impact of IRPC operations on biodiversity
- quarantine should be strengthened
- the impacts of external lighting should be monitored to assess whether the intensity and duration of night lighting need to be reduced
- weeds must be controlled
- stormwater controls need to be reassessed to prevent run-off into the national park
- wildlife mortality on roads should be monitored and minimised by, for example, using buses to transport staff.

Figure 1: Map of Christmas Island



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Coast, Roads and National Park data © Commonwealth of Australia, 2008

Caveats:
The information presented in this map has been provided by a range of groups and agencies. While every effort has been made to ensure accuracy and completeness, no guarantee is given, nor responsibility taken by the Commonwealth for errors or omissions, and the Commonwealth does not accept responsibility in respect of any information or advice given in relation to, or as a consequence of, anything containing herein. The map has been collated from a range of sources, with data at various resolutions. Data used are assumed to be correct as received from the data suppliers.

1. Introduction

Christmas Island is located in the Indian Ocean at 10° 25’S and 105° 40’E. The island covers approximately 135 square kilometres. It has a typical tropical equatorial climate with distinct wet and dry seasons, but with high year-round humidity. The dominant terrestrial habitats are rainforests that show subtle variations in canopy height and density and deciduousness depending on soil depth, altitude and aspect.

Christmas Island supports a wide range of unique and unusual species and habitats, and although it has been mined for phosphates for much of the past century, most of the natural ecosystem remains intact. Being so isolated, the island’s ecosystem is highly closed, with more than 200 endemic species and subspecies of plants and animals, including the largest and most diverse land crab community in the world. The declaration of the Christmas Island National Park placed more than 60 per cent of the island under formal protection.

The Christmas Island Biodiversity Monitoring Program (BMP) was funded by the Australian Government Department of Finance and Deregulation and implemented by Parks Australia between December 2003 and April 2007. The purpose of the program was to develop a baseline for the island’s biodiversity and to monitor the effects on biodiversity of constructing an Immigration Reception and Processing Centre (IRPC) at North West Point on Christmas Island.

The main objective of the BMP was to establish baseline data and then monitor trends in selected flora and fauna populations to determine what mitigating actions, if any, would be required following construction of the IRPC to prevent declines in any species or ecological processes. The BMP focused on collecting baseline data on as many groups of biodiversity as possible within the three-year timeframe. Another important goal was to integrate different studies of biodiversity where possible.

The program allowed Parks Australia to gain a much greater understanding of the breadth and significance of biodiversity on Christmas Island and the threats and challenges that exist in maintaining its integrity. The results from the program have provided a good foundation for building scientific knowledge, including databases, into the future. Although the BMP ended in April 2007, Parks Australia has continued to monitor key species.

1.1 Checklist of flora and fauna of Christmas Island

A checklist of the flora and fauna recorded on Christmas Island was initiated as part of the BMP. A spreadsheet database was developed and it is divided into three worksheets: plants, invertebrates and vertebrates. Each sheet contains the following fields:

- class
- order
- family
- subfamily
- scientific name

- taxonomic authority (the reference to the original scientific description)
- English name
- synonyms (which may include misidentifications, and invalid names)
- level of endemism (categorical: endemic, native or introduced)
- conservation status.

A total of 3,903 species is listed (483 plants, 2,667 invertebrates and 753 vertebrates). The list was compiled from existing literature, and museum reports. As with all such inventories, this is a work in progress.

1.2 Christmas Island biodiversity inventory database

Parks Australia developed a template for biodiversity inventories to be used across all Commonwealth reserves to meet the requirements of the *Environment Protection and Biodiversity Conservation Act 1999* (Cwlth) (EPBC Act). The template was adapted as part of the BMP to cover the conservation status of 576 species and subspecies of plants and animals of conservation significance on Christmas Island. These include 253 endemic taxa and 165 taxa occurring nowhere else in the Australian jurisdiction, and 110 species listed as threatened, migratory or marine under the EPBC Act.

A scoring system provided a way of ranking the taxa by conservation priority. The conservation status of this unique biodiversity is very poor. Nearly 70 species are missing (that is, they have not been recorded for decades) and the status of many others is unknown.

The Christmas Island biodiversity inventory database focused on native species of conservation significance. Introduced and invasive species were not covered. An inventory of invasive species on Christmas Island is required so that appropriate prioritisation and planning can be undertaken to mitigate existing and emerging threats to biodiversity.

2. Christmas Island pipistrelle

The Christmas Island pipistrelle *Pipistrellus murrayi* is a tiny insectivorous bat that is confined to Christmas Island. It is one of the smallest bats in the world.

The pipistrelle was evidently common from the time of settlement in 1895 until about 1988. In 1984, it was considered to be 'widely distributed on the island, in both terrace and plateau forest' and had a 'seemingly uncluttered distribution' across the island (Tidemann 1985, pp. 6 and 8). It was recorded foraging over the high school in Drumsite and feeding on insects inside the Christmas Island Club, which was located behind Flying Fish Cove. At that time, the population was considered to be stable and secure (Tidemann 1985). Pre-dusk activity was conspicuous.

By 1994, both abundance and distribution of the pipistrelle were patchy, which suggests that the species was already in decline, having undergone a marked reduction in distribution (Lumsden and Cherry 1997). In 1998, the pipistrelle range had undergone a considerable westward contraction since both 1984 and 1994 (Lumsden et al. 1999). During that study, 96 per cent of pipistrelle records came from the western third of the island. Between 1998 and 2004, the westward trend in range contraction continued.

In 2001, the pipistrelle was listed as *endangered* under the EPBC Act, and in September 2006, it was upgraded to *critically endangered* (the highest threat category available). It is the most endangered species of mammal in Australia, and one of the most endangered species of bats in the world. The reasons for the decline are not known.

Considerable and strategic research commitment is urgently required for the recovery of the Christmas Island pipistrelle population. Recommendations for future research directions are discussed in the context of urgency and the efficient use of resources.

2.1 Summary of the results

The surveys conducted as part of this BMP show that the pipistrelle has continued to decline in abundance, and its distribution has contracted to a small part of the island close to the IRPC (Figure 2). About 95 per cent of remaining pipistrelles feed in this area every night, and are therefore potentially affected by ongoing activities there.

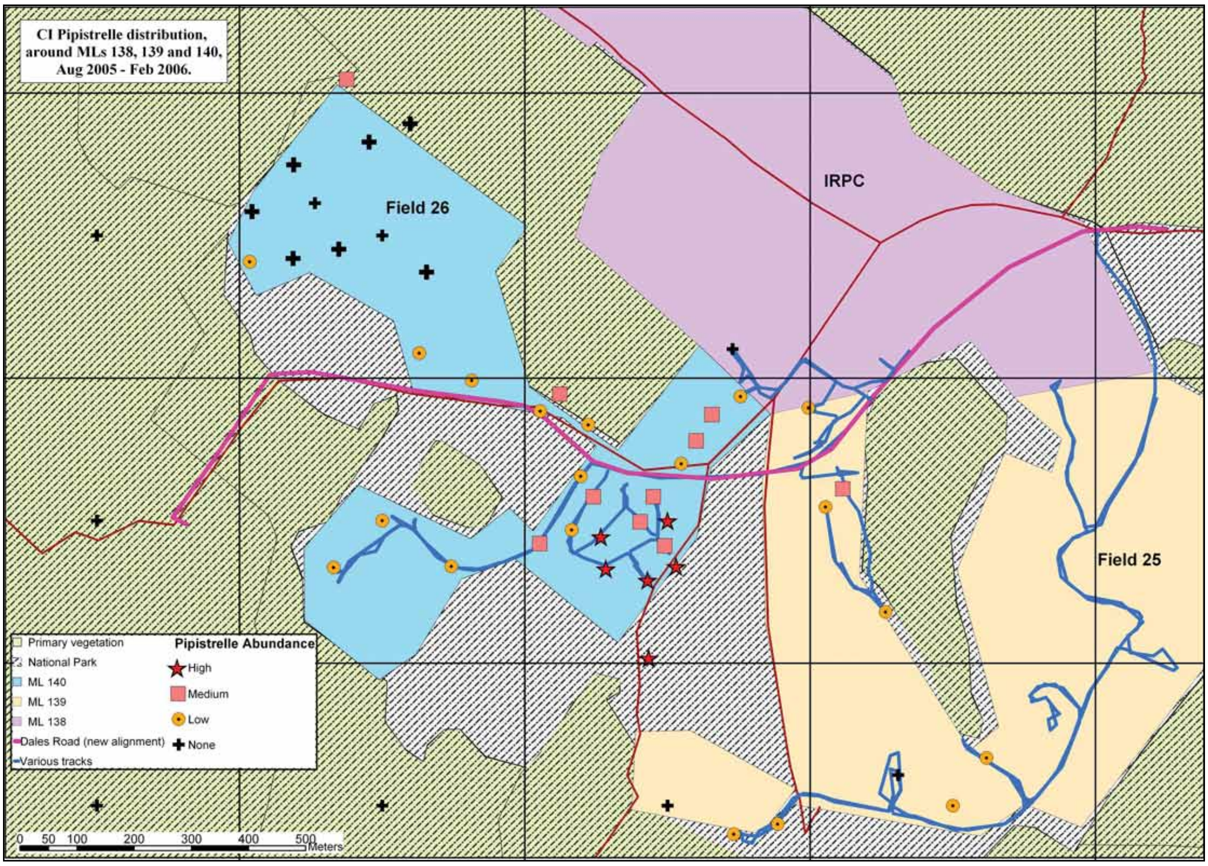
The pipistrelle population has declined substantially in the last decade. By the time of this study (2004 to 2006), the pipistrelle was recorded only in the Field 22 area of the central west (at low density), along the westernmost two kilometres of North West Point Road (at low density), in the Field 26 area, and in part of the western coastal terrace. This represents a substantial decline in the range of the pipistrelle over the 22 years from 1984 to 2006.

There is also strong evidence from these studies that foraging behaviour has changed, with the pipistrelle found to leave nests after dark and preferring to forage in secondary growth rather than in primary forest in 2004 and 2005.

The pipistrelle's habitat requirements are poorly known, but they formerly occurred in primary and secondary forests, ecotones and urban environments. Their diet is also poorly known, but there is evidence of flexibility and adaptability, and little evidence of extreme specialisation. We are reasonably certain that no single predator (including the yellow crazy ant *Anoplolepis gracilipes* and the wolf snake *Ophites aulicus capucinus*) is solely responsible for the decline. There is little evidence for fundamental ecosystem failure, such as a collapse of the food base or loss of habitat, being the cause of the decline. The possibility of a disease epidemic is highly plausible, although it is supported only by circumstantial evidence.

More detailed results are provided below.

Figure 2: Map of Christmas Island pipistrelle distribution and abundance in the vicinity of the IRPC during August 2005 and February 2006



2.2 Research and monitoring methods

Research results presented in this report were collected on Christmas Island between January 2004 and December 2006. The type of research can be broadly classified into three categories:

1. Mapping the species distribution
   - on broad scales (island-wide) in 2004
   - on fine scales in 2005.
2. Monitoring trends in the species' relative abundance at selected sampling sites
   - on broad scales (island-wide) in 2004 and 2005

  - on fine scales in 2006.
3. Assessing potential threats
   - studying some potential predators in 2004 and 2005
   - monitoring roost sites in 2006.

The raw data are stored as Microsoft Excel® spreadsheets on the computer network at Parks Australia Christmas Island. Geographical data and maps are also stored as Esri ArcMap® shape files, and map documents, and the map layouts are stored as 300 dpi tif images.

Details of the methods are in Appendix A.

2.3 Detailed survey and monitoring results

2.3.1 Broad-scale mapping of foraging sites and distribution in 2004 and 2005

Foraging surveys in 2004 and 2005 showed that the range of the pipistrelle had contracted significantly since Lumsden's survey in 1998 (Lumsden et al. 1999).

In 1998, pipistrelles were recorded in transect surveys from a moving vehicle at about 30 sites, mostly in the western and central sections of the island, with three sites in the south and one site in the east.

In 2004 to 2005, about 90 per cent of the area surveyed in 1998 was covered, plus an additional 20 per cent of the island which was not surveyed in 1998. Despite the greater coverage of the island and more careful surveying (slower driving speeds), bats were recorded at only seven sites: six in the far west and one in the centre of the island. These results indicate that both the range and abundance of the pipistrelle declined substantially between 1998 and 2005.

Specifically, the pipistrelle apparently vanished, at least largely if not completely, from the eastern, north-eastern, south-eastern and central sections of the island between 1998 and 2005 (Table 1).

Table 1: Results of 2004 and 2005 broad-scale transect surveys

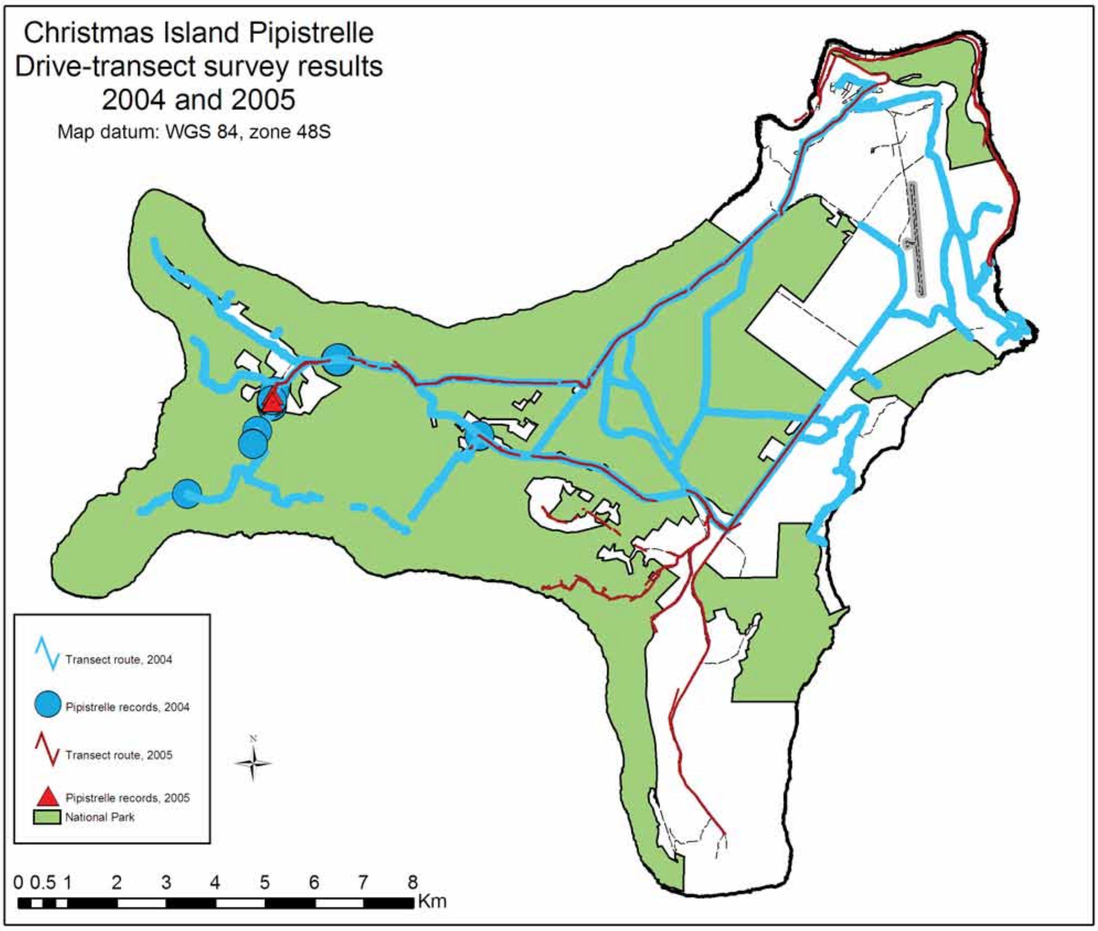
2004 survey	**2005 survey**
Pipistrelles were detected at only seven locations: • five sites along the Winifred Beach Track • one site on North West Point Road, about one kilometre to the east of the IRPC • one site in Field 22 at the corner of the East–West Baseline and the Aldrich Hill (Circuit) Track.	Pipistrelles were detected at only two locations in total, namely: • two sites along the Winifred Beach Track.
No pipistrelles were recorded in the eastern or north-eastern sections of the island, and with the exception of the site in Field 22, none were recorded in the central section.	No pipistrelles were recorded in the eastern, north-eastern, south-eastern or central sections of the island.

Quantitative comparisons need to be treated with caution, because the survey effort and methodology varied between surveys. Nevertheless, between 1998 and 2004–2005, there was:

- a decline of 88 per cent in individual records (from 99 to 12)
- a decline of 77 per cent in the number of specific sites that pipistrelles were recorded at (from 30 to seven)
- and a decline of 74 per cent in the number of 100-metre grid squares in which pipistrelles were recorded (from 19 to five).

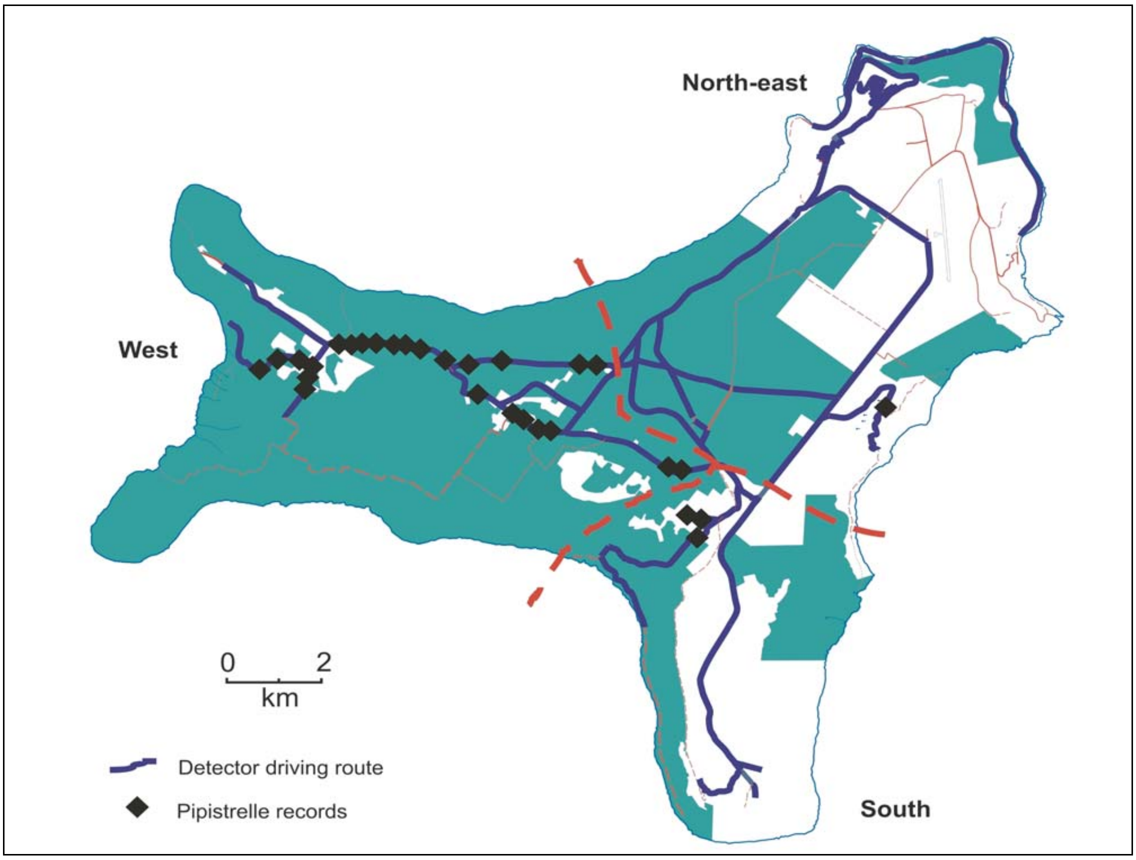
Both the 2004 and 2005 transect survey results are shown in Figure 3. The 1998 transect survey details are shown for comparison in Figure 4 (reproduced from Lumsden et al. 1999, courtesy of L. Lumsden).

Figure 3: Results of drive-transect surveys for Christmas Island pipistrelle in 2004 and 2005



Short gaps in the transect lines as mapped are the result of incomplete GPS coverage, not actual breaks in the transects

Figure 4: Results of drive-transect surveys for Christmas Island pipistrelle in 1998



Reproduced from Lumsden et al. (1999), courtesy of L. Lumsden. The red dashed lines were used for geographical analyses by Lumsden et al. They are not used in this report.

A second series of surveys using stationary detectors at foraging sites confirmed the observations, with data showing the range of the pipistrelle had contracted during the past seven years, from an area in the western part of the island of 60 square kilometres to only 20 square kilometres. The abundance of pipistrelles also declined (Table 2), with the average decline over seven years considered significant ($\chi 2 = 27.68$; df=2; $P<0.0001$). The initial sites surveyed by Lumsden et al. (1999) are known as ‘Lumsden’ sites.

Table 2: Comparison of pipistrelle abundance at ‘Lumsden’ monitoring sites between 1998 and 2005

Measure of relative abundance	1998	2004	2005
Number of ‘Lumsden’ sites re-surveyed	51	43	35
Number of sites ranked ‘high’ activity	4	2	2
Number of sites ranked ‘medium’ activity	13	1	0
Number of sites ranked ‘low’ activity	17	6	5
Number of sites ranked ‘no’ activity	17	34	28
Total number of sites with bats	34	9	7
Total number of bat calls*	2,497	1,023	974

1998 data from Lumsden et al. (1999) and 2004 data from this BMP;
*bat calls recorded from 18:30 to 21:00 in 1998 and from 18:00 to 21:00 in 2004 and 2005.

2.3.2 Fine-scale mapping of foraging sites and behaviour

A new sampling design was developed to monitor the population and distribution levels at a finer scale than in the 2004 and 2005 surveys, as the continued population decline meant that most of the ‘Lumsden’ sites established in 1998 no longer contained pipistrelles. Details of the methods used are in Appendix A.

Foraging behaviour

A shift in the diel activity pattern of pipistrelles occurred between 1984 and 2004. Pre-dusk activity was conspicuous in the 1980s, but noticeably absent in 2004. It also appears that the post-dusk activity was sustained longer into the night in 2004 than in the 1980s. Data from 1994 and 1998 indicate that the shift in diel activity may have been gradual and correlated with the declines in distribution and abundance.

In 1984, pipistrelles were regularly observed foraging in the afternoons, as early as 1.5 hours before sunset (6 pm), and most often around forest edges and along tracks known as drill lines that covered the island at the time (Tidemann 1984). In 1994, Lumsden and Cherry (1997) observed a shift, with moderate activity in the hour before sunset, a very high activity peak in the hour directly after sunset, a return to moderate activity from 7 pm until midnight, and low activity after midnight with the exception of a moderate peak just before dawn.

Between 2004 and 2006, pipistrelles were not recorded leaving roost sites until about half an hour after dusk, and they were not recorded on detectors in foraging areas until half an hour after dark. We cannot explain the preference for foraging in secondary growth that was recorded between 2004 and 2006. Pipistrelles mostly continue to forage along edges and around surfaces of vegetation, as they did in 1984 (Tidemann 1985) and 1994 (Lumsden and Cherry 1997). They do not forage in extensively open and cleared areas, but occasionally commute through them.

By 1994, most of the drill lines had regrown. The closure of the drill lines may have led to a loss of foraging habitat and this could account for the species decline (Corbett et al. 2003); but it does not explain how the pipistrelle was abundant before any forest disturbance or why this study observed no geographical correlation between the closure of drill lines and westward range contraction. An increase in abundance of the potential predator nankeen kestrel *Falco cenchroides* provides the only apparent explanation for this change, although this has not been investigated in detail.

Fine-scale mapping of foraging distribution in 2005

The fine-scale survey of 63 sites (30 interior and 33 exterior) identified that, in 2005, probably more than 80 per cent of pipistrelles foraged in a small area around mining leases ML140 (Field 26), ML139 (Field 25), ML138 (where the IRPC is located) and adjacent areas of national park.

The centre of the foraging was at the top of the Winifred Beach Track where the only ‘high’ levels of activity were recorded. However, the foraging area extended along parts of the Dales Road, throughout most of Field 26, to parts of Field 25, around the edges of the IRPC, and along the North

West Point Road one to two kilometres eastwards from the IRPC (Figure 5). Abundance generally declined with increasing distance from the top of the Winifred Beach Track, although habitat is likely to have had a strong influence on this.

There were generally more pipistrelles in exterior or edge sites than in interior sites (Table 3), with the difference being significant ($\chi 2= 34.85$; df=1; $P<0.0001$). In 2005, the remaining ‘primary’ foraging sites were in secondary habitats away from ecotones and roads. The pipistrelles appeared to be foraging mostly in secondary regrowth (containing a mixture of native vegetation and weeds), and using tracks and clearings as flying lanes. Pipistrelles were generally absent from extensively open and cleared areas.

Figure 5: Results of fine-scale mapping and monitoring surveys in the west of the island for Christmas Island pipistrelle in 2005

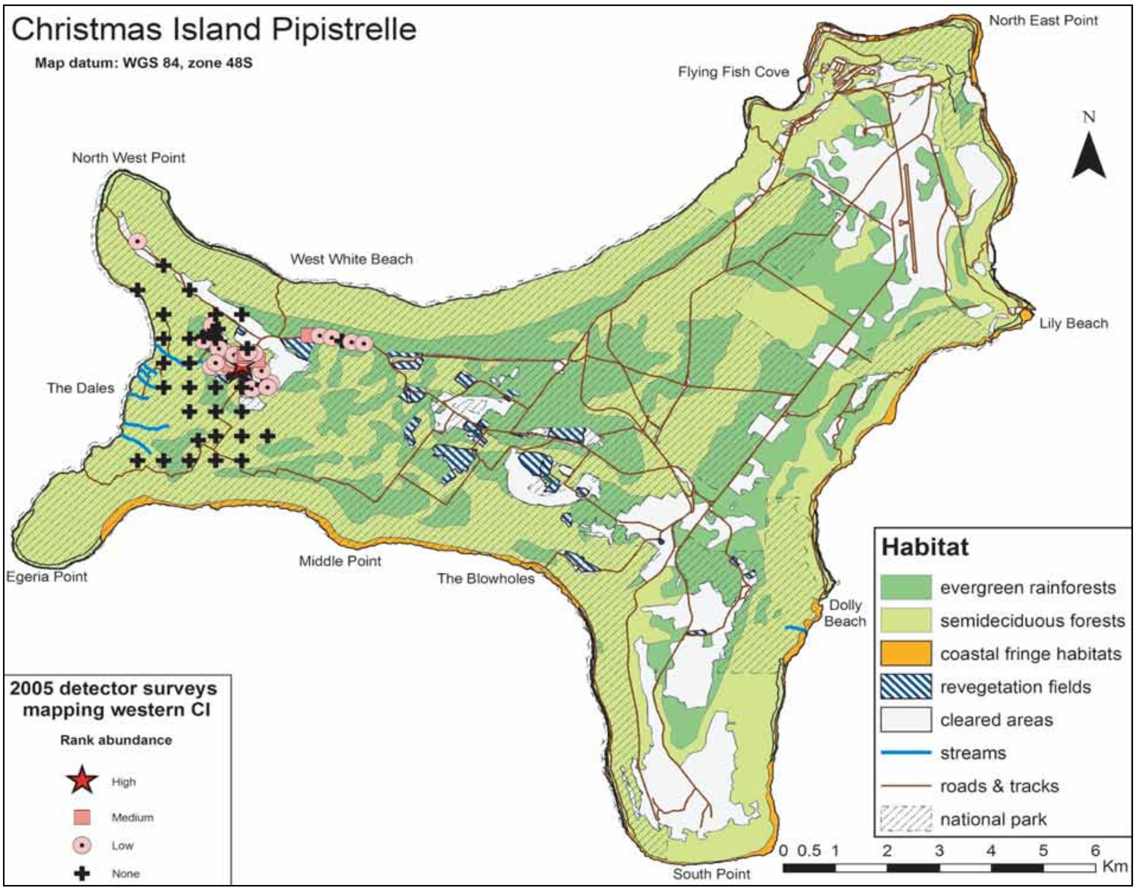


Table 3: Summary of 2005 pipistrelle bat detector surveys in the west of Christmas Island, classified by habitat and rank abundance

Habitat	Total number of sites	Number of sites ranked High	Number of sites ranked Medium	Number of sites ranked Low	Number of sites ranked None
Interior	31	1	0	2	28
Exterior	46	3	9	24	10

Interior habitats defined as sites inside primary forest away from clearings, roads and tracks. Exterior habitats defined as sites in secondary habitats along roads and tracks, and in clearings.

Fine-scale monitoring of foraging sites in 2006

The locations of the 12 sites chosen for repeated fine-scale monitoring with bat detectors during 2006 are shown on Figure 6. During 2006, 513 nights of monitoring data were collected from the 12 sites. Additionally, available data were compiled from a further 75 nights in 2004 and 2005 for some of the sites to increase the temporal span of the data set. Table 4 summarises the results from these sites for 2004 to 2006, and Table 5 breaks up the 2006 data by quarters.

Over the course of the year, the relative abundance of pipistrelles declined significantly at one of the 12 sites (A3), increased significantly at three sites (A4, D03 and Z03), and did not change much at eight sites. The overall trend across the 12 sites was one of significant decline because the decline in abundance at site A03 was an order of magnitude greater than the increases at A4, D03 and Z03. The results revealed a highly significant decline in the overall abundance of bats foraging at the 12 sites during 2006. The large variation within sites makes interpretation of the data complex. It is most likely that many of the bats were simply foraging in slightly different areas on different nights.

It is apparent from the data that, as abundance declined in the core of the Field 26 foraging area (A sites), it increased in the periphery of the Field 26 foraging area (Z sites). Since these six sites are within one kilometre of each other, it is almost certain that at least some of the bats were foraging at multiple sites. Meanwhile, abundance remained constantly low at the outlying (R and S) sites.

Figure 6: Summary of stationary detector sites where pipistrelles were recorded in 2004–05, and showing the location of the 2006 fine-scale monitoring sites

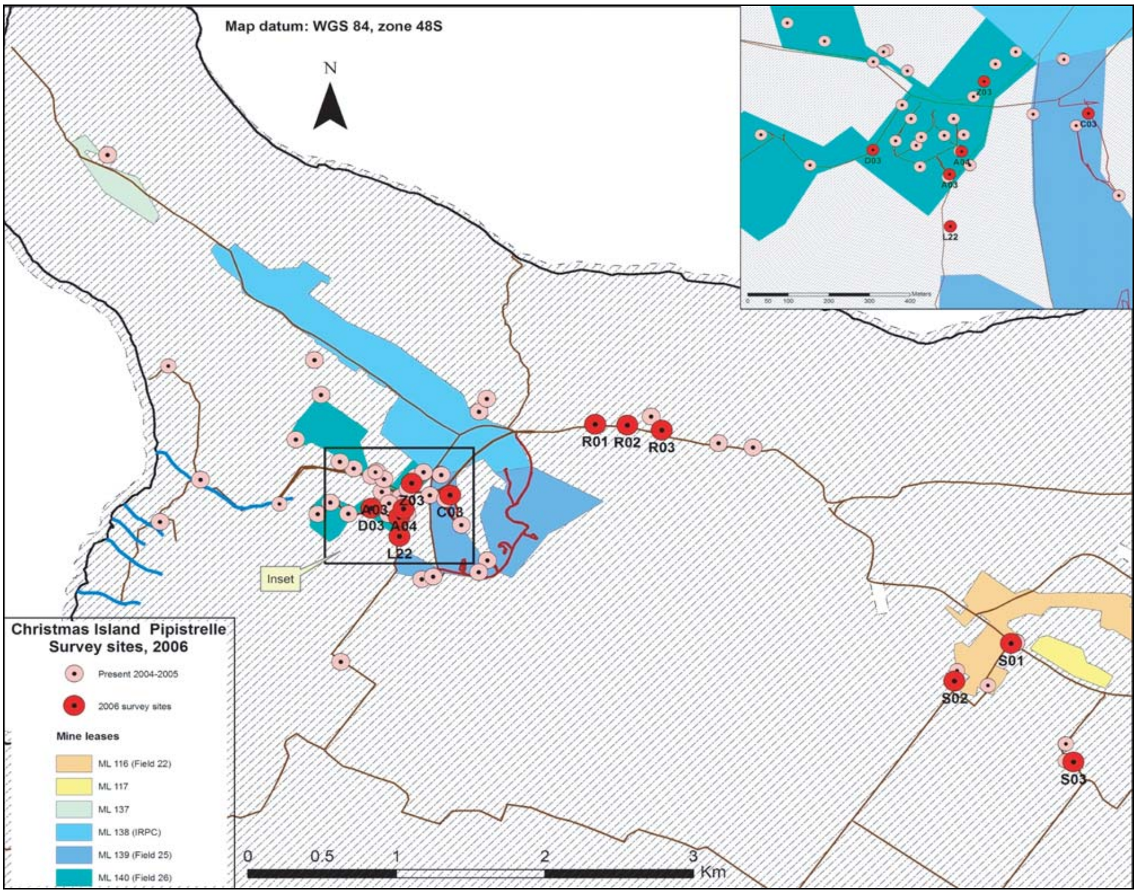


Table 4: Summary statistics for the 12 fine-scale monitoring sites, 2004–2006, by year

Period		A sites			R sites			S sites			Z sites			All sites
		A3	A4	L22	R01	R02	R03	S01	S02	S03	C03	D03	Z03	
2004	nights	0	0	3	0	0	0	1	0	0	0	0	0	4
	total			1,552				218						1,770
	mean			517				218						443
	sd			349										322
	min			192				218						192
	max			886				218						886
2005	nights	7	11	43	1	1	5	0	0	0	1	1	1	71
	total	8,325	851	13,119	75	16	8				191	158	198	22,941
	mean	1,189	77	305	75	16	2				191	158	198	323
	sd	463	95	199			3							322
	min	253	8	5	75	16	0				191	158	198	0
	max	1,655	349	720	75	16	6				191	158	198	1,655
2006 all	nights	34	39	43	42	50	56	48	39	42	45	33	42	513
	total	24,797	1,421	8,976	691	563	166	1,291	29	652	2,918	3,316	2,615	47,435
	mean	729	36	209	16	11	3	27	1	16	65	100	62	92
	sd	436	70	251	26	32	6	62	2	36	168	87	89	322
	min	8	0	6	0	0	0	0	0	0	0	7	0	0
	max	1,947	315	1,013	164	214	36	318	11	164	856	281	320	1,947
2004 – 2006	nights	41	50	89	43	51	61	49	39	42	46	34	43	588
	total	33,122	2,272	23,647	766	579	174	1,509	29	652	3,109	3,474	2,813	72,146
	mean	808	45	266	18	11	3	31	1	16	68	102	65	123
	sd	469	77	237	27	32	6	67	2	36	167	86	90	322
	min	8	0	5	0	0	0	0	0	0	0	7	0	0
	max	1,947	349	1,013	164	214	36	318	11	164	856	281	320	1,947

Nights = number of nights of data collected; total = total number of calls recorded in the period; mean = mean number of calls per night for the period; sd = standard deviation of the mean; min = minimum number of calls recorded in a night; max = maximum number of calls recorded in a night

The data were collected during one year and exactly between successive breeding seasons (breeding occurs in December to January each year; Lumsden et al. 2006). On 16 January 2007, an emergence count at roost tree 14 found that some bats remained in the roost site (L. Barrow and M. Bramson, pers. comm.; Figure 7). These bats were most likely juveniles that were not yet flying. Therefore, the 2006 detector data set probably began about the time that the 2006 cohort was joining the population but ended before the 2007 cohort joined. It thus includes one year of juvenile mortality but did not record the juveniles that joined the foraging population after the preceding and following breeding seasons. With only one year of data it is not possible to accurately partition juvenile mortality from the overall population decline.

In the absence of further information, the decline during 2006 at the fine-scale monitoring sites can be estimated at 49 per cent, which includes the mortality of the 2006 cohort. The fine-scale monitoring sites sampled all of the known, remaining foraging range of the pipistrelle in 2006, and half the sites were in the core foraging area at Field 26. These results provide evidence of a significant decline in the number of pipistrelles around the IRPC.

Note that ‘A’ sites decreased, the nearby ‘Z’ sites increased, and the outlying ‘R’ and ‘S’ sites remained constantly low. Note also that the declines at ‘A’ sites were an order of magnitude greater than the increases at the ‘Z’ sites.

Table 5: Summary statistics for the 12 fine-scale monitoring sites in 2006, by quarter

Period		A sites			R sites			S sites			Z sites			All sites
		A3	A4	L22	RO1	RO2	RO3	SO1	SO2	SO3	CO3	DO3	ZO3	
1st quarter (Jan to Apr)	nights	10	11	12	9	8	13	12	8	9	12	10	10	124
	total	10,193	458	4,029	285	31	36	83	18	29	88	272	42	15,564
	mean	1,019	42	336	32	4	3	7	2	3	7	27	4	126
	sd	447	54	440	50	4	4	11	4	6	8	21	6	322
	min	202	0	6	4	0	0	0	0	0	0	8	0	0
	max	1,947	146	1,013	164	14	11	35	11	17	22	75	18	1,947
2nd quarter (Apr to June)	nights	8	11	6	11	11	9	9	8	4	6	8	6	97
	total	6,908	72	915	84	397	16	67	2	327	172	499	0	9,459
	mean	864	7	153	8	36	2	7	0	82	29	62	0	98
	sd	351	9	110	7	64	2	15	0	94	43	63	0	322
	min	500	0	18	0	0	0	0	0	0	0	7	0	0
	max	1,556	25	312	24	214	6	45	1	164	110	167	0	1,556
3rd quarter (July to Sept)	nights	13	13	16	20	25	16	16	16	22	14	7	22	200
	total	7,667	183	2,158	316	91	28	1,135	9	296	2,580	1,035	2,542	18,040
	mean	590	14	135	16	4	2	71	1	13	184	14	116	90
	sd	260	23	83	15	5	5	93	2	18	268	94	95	322
	min	186	0	10	0	0	0	0	0	0	2	48	0	0
	max	960	82	245	57	15	18	318	7	76	856	281	320	960
4th quarter (Oct to Dec)	nights	3	4	9	2	6	18	11	7	7	13	8	4	92
	total	29	708	1,874	6	44	86	6	0	0	78	1,510	31	4,372
	mean	10	177	208	3	7	5	1	0	0	6	189	8	48
	sd	3	132	80	1	9	10	1	0	0	10	45	13	322
	min	8	0	95	2	0	0	0	0	0	0	114	0	0
	max	13	315	298	4	24	36	4	0	0	29	248	27	315

Nights = number of nights of data collected; total = total number of calls recorded in the period; mean = mean number of calls per night for the period; sd = standard deviation of the mean; min = minimum number of calls recorded in a night; max = maximum number of calls recorded in a night

Figure 7: Roost site under exfoliating bark on roost tree 14 in Sydneys Dale

Note the bats between the bark and the trunk. These are possibly young bats that cannot fly.
Photo: Mark Branson, 16 January 2007

2.3.3 Pipistrelle roosting sites

In addition to the results of this BMP, a number of recent reports have reviewed information in attempts to identify the cause of the continuing decline of the pipistrelle population (Lumsden et al. 1999; Schulz and Lumsden 2004). In 2005, Natural Heritage Trust funds were awarded to the Arthur Rylah Institute for Environmental Research (ARI) (of the Victorian Department of Sustainability and Environment) to test two leading hypotheses regarding threats:

- predators attacking pipistrelles at their roost sites
- disease or parasite loadings in the population (see Lumsden et al. 2006).

Working collaboratively with Parks Australia, Lindy Lumsden led a team from ARI to visit the island in December 2005 to trap pipistrelles, assess the health of the population, and radio-track individuals to locate their roost trees. Further details about the methods used are given in Appendix A.

Seven colonial (maternity) and three individual roost sites were located. Six of the maternity roost sites were located under the exfoliating bark on the trunks of dead trees (*Tristiropsis acutangula* and *Inocarpus fagifer*), and one was located in the top of the hollow trunk of a dead palm (*Arenga listeri*). Figure 7 shows the roost site under exfoliating bark on roost tree 14.

Monitoring of the above roost trees revealed that by May 2006 four of them had fallen down and one was no longer in use (Lumsden et al. 2006; BMP data). Radio-tracking of nine females identified two additional roost trees. In addition, one of the roost trees located in December 2005 was also found to be still active (Hoye 2006).

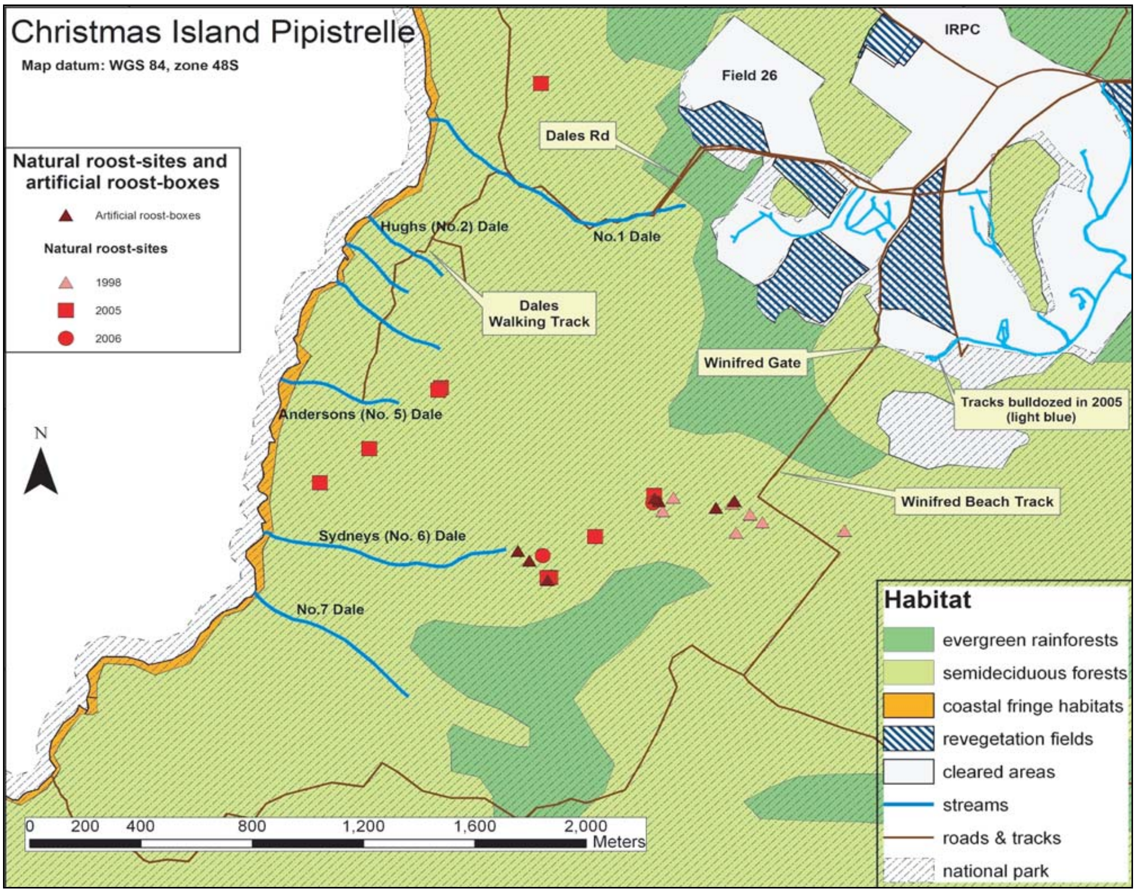
Table 6 summarises the characteristics of pipistrelle roost sites. The majority of natural roost sites are in Sydneys Dale below its junction with the Winifred Beach Track (Figure 8). Five other roost sites were located in 1998, and one in 1984 for which no coordinates are available. This includes one roost tree located near Jacks Hill, the only known roost site outside the Dales area (see Lumsden et al. 1999).

Table 6: Summary of pipistrelle roost-site characteristics

Roost-habitat	1998	2005	2006	Total
Live tree	2 colonial	0	0	2 colonial
Dead tree	6 colonial 1 unknown	6 colonial	2 colonial	14 colonial 1 unknown
Live Arenga palm	1 colonial 1 single	1 single	0	1 colonial 2 single
Dead Arenga palm	0	1 colonial	0	1 colonial
Live Pandanus	1 single	2 single	0	3 single
Total	9 colonial 2 single 1 unknown	7 colonial 3 single	2 colonial	18 colonial 5 single 1 unknown

Data from Lumsden et al. (1999, 2006) and Hoye (2006)

Figure 8: Locations of natural roost sites and artificial roost boxes



<u>Monitoring the occupancy of roost sites in 2006</u>

In early 2006, the evening emergence of pipistrelles was counted at three communal roost sites. A total of 80 bats was counted emerging from the three sites. Two of the roost trees (13 and 23) fell down shortly after these emergence counts. A lack of known roost trees at which to count emergences and a lack of resources meant that very little emergence data were collected by direct counts. Table 7 summarises all available count data for emergences of pipistrelles from communal roost sites, including counts from 1998 and 2005 (from Lumsden et al. 1999, 2006).

Four roost trees were surveyed for the presence of roosting bats using bat detectors in 2006. Detectors were left underneath roost trees for periods of one to two weeks. The detectors did not count the number of bats in the roost sites, but were used to determine whether the roost sites were occupied. Bats emerged from roost sites between about 6:15 pm and 6:45 pm (Table 7) and returned at about 5:30 am. When the roost sites were occupied, the detectors recorded two peaks of activity just after dusk and just before dawn, with a lull in the middle of the night. More information about the use of bat detectors is in Appendix A.

Overall, the results suggest that the lack of roosting sites might be an issue for the pipistrelle population.

Table 7: Counts of evening emergence of pipistrelles from communal roost sites

Date	Roost no.	Observer	First emerge	Last emerge	Bat count
6/06/1998	1	ARI			47
7/06/1998	2	ARI			13+
8/06/1998	4	ARI			41
9/06/1998	5	ARI			28+
11/06/1998	8	ARI			6
12/06/1998	10	ARI			1
13/06/1998	6	ARI			45
15/06/1998	11	ARI			1
16/06/1998	6	ARI			31
14/12/2005	6	ARI			0
15/12/2005	13	ARI	6:18 pm	6:40 pm	35
16/12/2005	13	ARI	6:26 pm	6:46 pm	53
16/12/2005	14	ARI	6:19 pm	6:29 pm	32
17/12/2005	15	ARI	6:23 pm	6:44 pm	48
18/12/2005	13	ARI and BMP	6:29 pm	6:41 pm	54
20/12/2005	17	ARI	6:23 pm	6:36 pm	12
22/12/2005	18	ARI	6:18 pm	6:39 pm	14
24/12/2005	17	ARI	6:18 pm	6:31 pm	15
30/12/2005	23	ARI	6:41 pm	6:54 pm	11
30/12/2005	21	ARI	6:27 pm	6:41 pm	39
31/12/2005	13	ARI	6:27 pm	6:53 pm	52
18/01/2006	23	BMP	6:44 pm	6:59 pm	15
15/03/2006	21 and 14	BMP	6:16 pm	6:30 pm	47
16/03/2006	13	BMP	6:15 pm	6:31 pm	18

1998 data from Lumsden et al. (1999) and 2005 data from Lumsden et al. (2006)

2.4 Assessment of threats

The cause or causes of the pipistrelle's decline remain unknown, despite the efforts made to identify them. Possible factors include: introduction of predators or diseases through inadequate quarantine, inappropriate land management, direct persecution and climate change.

Lumsden et al. (2006) investigated disease and parasite loadings in the population and found no evidence for either. They also laid the groundwork for the monitoring of predators at roost sites, as this has emerged as a likely cause of past declines. Loss of habitat is possibly the most serious threat over the next few years, although this was unlikely to have caused the initial declines.

2.4.1 Potential predators

Four cameras were deployed at five roost sites and two control sites, for a total of 663 trap nights between April and December 2006. Huntsman spiders *Heteropoda venatoria* were the most frequently recorded animals on the trunks of the roost trees, with 43 individual records. Table 8 summarises the survey effort and findings from the remote cameras up to the end of December 2006. A wolf snake *Ophites aulicus capucinus* was photographed on roost tree 565 on 1 February 2007, after the cut-off date of this report.

Table 8: Summary of photographs of potential predators near pipistrelle roost sites

Roost number	**6**	**13**	**14**	**17**	**21**	**565**	**686**	**A**	**Total**
Days deployed	28	22	158	135	98	97	113	12	663
Days with photographs	26	19	41	65	48	46	77	6	328
Number of photographs	982	100	682	1,898	106	1,490	760	145	6,163
Photographs with nothing significant detected	982	100	618	1,784	83	996	748	145	5,456
Rats	0	0	0	4	1	0	0	0	5
Centipedes	0	0	0	7	1	0	0	0	8
Spiders	0	4	8	14	7	4	6	0	43
Crabs	0	0	0	3	0	0	1	0	4
Other invertebrates	0	0	6	6	12	3	3	0	30
Ants	0	0	3	0	1	7	0	0	11

Roost-sites 6 and A were control sites with no recently active bat roosts; counts of ants (*Camponotus* spp.) are the number of nights in which ants swarmed on the trunks; counts of other animals are estimates of the number of individuals captured on one or more photographs. 'Other invertebrates' include cockroaches, moths, crickets and beetles.

Native invertebrates including robber crabs *Birgus latro*, red crabs *Gecarcoidea natalis*, and huntsman spiders are not considered to be serious threats for a range of reasons. Firstly, they are native and have coexisted with the pipistrelle for thousands of years. Pipistrelle roost sites are outside the general foraging habitat of the crabs. Pipistrelles are likely to be well above the prey size-range of huntsman spiders. In addition, the cameras did not actually record any of these animals preying on bats, disturbing them or entering their roost sites.

This leaves three potential predators at roost sites to be considered seriously: (1) the giant centipede *Scolopendra morsitans*; (2) the black rat *Rattus rattus*; and (3) the wolf snake.

Giant centipedes have been identified as a possible predator (Hoye 2006; Lumsden et al. 2006). They were photographed on roost tree 17 seven times and on roost tree 21 once. Giant centipedes have been shown by the BMP to be abundant throughout the forests of Christmas Island. They have the voracity and the abundance to be one of the main causes of decline. However, centipedes were introduced to Christmas Island soon after settlement, were abundant by 1907 and distributed island-wide by 1939 (Andrews 1900, 1909; Gibson-Hill 1949). Therefore, there is a poor temporal relationship between the introduction of the centipede and the decline of the pipistrelle (which apparently began between 1984 and 1994 (Tidemann 1985; Lumsden and Cherry 1997)). It is possible that giant centipedes underwent a substantial population increase sometime in the 1980s or so, but there are no data available to confirm this.

The photos from tree 17 mostly show a centipede of about the same size and in the same position on several occasions, suggesting that the animal probably lived under the bark of the tree near the infrared sensors. During reptile surveys by the BMP in 2004, giant centipedes were frequently recorded inside logs and under bark. It is likely that many dead trees on Christmas Island, including pipistrelle roost trees, have giant centipedes living in them.

Rats have been implicated in the decline of insectivorous bats in New Zealand (Pryde et al. 2005). Black rats were recorded on tree 17 four times and tree 21 once. The images indicate that black rats are capable of scaling the trees even in the absence of bark, epiphytes and trailing vines. The distribution and abundance of black rats were last assessed in the 1980s (Tidemann et al. n.d.). During the duration of the BMP, black rats were regularly observed in vegetation during spotlighting activities, and occasionally during the day, indicating that they occur throughout the island and are very common if not abundant. Black rats evidently arrived on Christmas Island in 1899 (Andrews 1909; Pickering and Norris 1996). Therefore, the temporal relationship between the introduction of the black rat and the decline of the pipistrelle is poor. It is possible that, like giant centipedes, black rats underwent a substantial increase in abundance or distribution sometime in the 1980s or so, but there are no data available to confirm this.

A single wolf snake was photographed on roost tree 565 in February 2007. The wolf snake may be a potential predator as its arrival coincided with the start of the decline in the pipistrelle (Lumsden et al. 1999).

The capabilities of these three predators to move through the forest canopy (i.e. move from tree to tree without returning to the ground) has not been specifically assessed. It is likely that the giant centipede has much smaller foraging ranges than either black rats or wolf snakes, which would reduce its capabilities to travel through the canopy. Conversely, once they have reached the roost sites, giant centipedes are likely to access them more easily.

The potential for predators preying on pipistrelles away from roost sites has not been examined in detail.

2.4.2 Loss of foraging habitat

The *National recovery plan for the Christmas Island pipistrelle (*Schulz and Lumsden 2004) recommended that actions be taken to increase protection of known and potential pipistrelle habitat outside the national park and to assess the potential impact of phosphate stockpile removal. During the mapping of foraging areas in mid-2005, a number of freshly bulldozed tracks were discovered in secondary regrowth in mining leases ML140 (Field 26), ML139 (Field 25) and ML138. Bat detectors were used to survey these tracks, and it was found that relatively high numbers of pipistrelles were foraging in these areas.

It was later found that Phosphate Resources Limited (PRL; trading as Christmas Island Phosphates) had opened these tracks to survey phosphate resources on the leases for export and to supply the IRPC with topsoil and garden mulch. Parks Australia initiated discussions with PRL, the Department of Finance and Deregulation (DOFA), the then Department of Immigration, Multicultural and Indigenous Affairs (DIMIA), Baulderstone Hornibrook (lead contractors in the construction of the IRPC) and Fink Projects (IRPC Project Managers).

Earthworks for phosphate mining at Field 26 (ML140) began in October 2005, within 300 metres of the centre of the main foraging area of the pipistrelles. It is estimated that 10–20 hectares of secondary regrowth, including native vegetation and weeds, were cleared in the north-western corner of Field 26. After further consultations with the above parties, the PRL contracts to supply topsoil and mulch to the IRPC were amended to prevent the materials being sourced from MLs 140, 139 and 138, and PRL suspended operations in Field 26. Late in 2006, a decision was made that PRL would source soil and mulch from MLs 116 and 117 (Fields 22 and 23) instead. Parks Australia advised all parties that this area was also one of the three known foraging areas of the pipistrelle. However, works commenced in October 2006 to clear secondary vegetation and mine stockpiles in Fields 22 and 23 (Figure 9 and Figure 10).

A bat detector stationed at fine-scale monitoring site S01, in Field 22 within 100 metres of the mining works, recorded an average of one call per night over 11 consecutive nights in October 2006 (after mining commenced) compared with 111 calls per night on nine consecutive nights in August 2006. It is not known whether this is a direct impact of mining, but it seems likely. Plates 10 and 11 show areas of pipistrelle foraging habitat in Fields 22 and 23 that have been extensively cleared of vegetation since October 2006. The topsoil was delivered by PRL to a laydown area in ML138 at the front of the IRPC. Areas of secondary vegetation were cleared for this purpose (Figure 11 and Figure 12). The area is part of the only substantial remaining foraging habitat of the pipistrelle (as mapped in 2005), and PRL and the IRPC stakeholders were advised that clearing could impact upon the pipistrelle. The changes in vegetation cover at the laydown area are shown in Figure 9 (October 2006) and Figure 10 (February 2007). During 2006, Baulderstone Hornibrook carried out extensive sampling in ML138 (directly across the Dales Road from the IRPC site and under ‘Helicopter Hill’) for topsoil quantities and qualities, despite being advised that this area should not be cleared because it was pipistrelle habitat. On 4 February 2007, a few hectares of secondary regrowth were cleared by PRL in ML138 at this site (Figure 13 and Figure 14). The vegetation from that site was not pushed into windrows, so it was evidently mulched. PRL complied with a request to stop clearing. In March, PRL returned to the site and stockpiled topsoil for sale to Baulderstone Hornibrook and the Australian Government for the IRPC grounds.

The Dales Road runs through the foraging area at MLs 140, 139 and 138. In 2006 the road was widened and vegetation along its edges was cleared on several occasions. This included widening of the road in preparation for supplying topsoil for the IRPC, and burying of an earthing strip to protect IRPC telephone cables from potential electrical currents that might occur as a result of powerlines servicing the IRPC.

Large areas of the IRPC construction site were covered with weedy secondary growth, which was cleared during the early earthworks phase in 2002. This clearance occurred before the commencement of the BMP. At the time it was not known that the remnant pipistrelle population was using this type of vegetation as a foraging habitat, and it was considered to be of low conservation value. Given the proximity of the IRPC to the Field 26 foraging area, it is highly likely that pipistrelles used parts of the IRPC site as a foraging area prior to 2002. However, no data are available to assess this.

It is not known why the pipistrelle, a species that evolved in a closed-forest environment, should now appear to prefer to forage in secondary regrowth. There are only three areas that the remnant population is known to use for foraging: Field 26, and surrounds; North West Point Road just east of the IRPC; and Fields 22 and 23. Habitat in all three of these areas has been extensively damaged as a result of the supply of utilities (power, water and phone infrastructure), topsoil and mulch to the IRPC. PRL has also directly contributed to habitat clearance, both for IRPC purposes (clearing vegetation to supply soil and mulch) and their own purposes (clearing vegetation to mine). At the time of writing, PRL still intends to mine extensive areas of Field 26.

It should be noted that three objectives in the recovery plan are relevant to land management in the vicinity of the IRPC and Field 22:

- Objective 9: Increase protection of habitat outside the national park
- Objective 10: Assess potential impact on pipistrelles of stockpile removal within and abutting the national park
- Objective 11: Develop guidelines to reduce road mortality.

With the population in rapid decline, preservation of all known foraging habitat of the pipistrelle is essential for the survival of this species.

Figure 9: Pipistrelle foraging habitat at Field 22 (100 metres from monitoring site S01), on 5 February 2007, cleared of secondary vegetation to provide mulch for the IRPC gardens

Figure 10: 200 metres from monitoring site S01, on 5 February 2007, cleared of secondary vegetation to provide mulch and topsoil for the IRPC gardens

Figure 11: IRPC construction site on 13 October 2006, showing remnant, secondary regrowth in the foreground

Figure 12: IRPC construction site on 5 February 2007, topsoil stockpiles where secondary regrowth formerly provided pipistrelle foraging habitat

Figure 13: Pipistrelle foraging habitat in ML138 (opposite the IRPC) on 5 February 2007, cleared of secondary vegetation

Figure 14: Pipistrelle foraging habitat in ML138 (opposite the IRPC) on 5 February 2007, cleared of secondary vegetation

Any impacts or changes to the foraging habitat of this species could cause further declines in population numbers. The few remaining areas of foraging habitat are now critically important to the species' survival. Clearance of vegetation in the foraging area at Field 26 has the potential to lead to the imminent extinction of the pipistrelle. However, it is not possible to predict with certainty what could happen. The pipistrelles might move to new foraging grounds and survive.

The only spots where pipistrelles can now be caught are in Field 26, and it is necessary to catch them so that their roost sites can be located and conservation management can be implemented. Irrespective of this, the precautionary principle must be applied: loss of foraging habitat could lead to further population decline. Clearance of Field 26 could leave the species with virtually no hope of avoiding extinction. Persistent damage on smaller scales (clearing a bit here and there) will ultimately have the same effect.

2.4.3 Loss of roosting habitat

The single communal roost discovered in 1984 was in a live tree, as were about half the roost trees discovered in 1998. All the communal roost trees discovered in 2005 and 2006 were in dead trees.

Table 7 summarises the characteristics of roost sites found in 1998, 2005 and 2006. Most communal roost sites that have been found were under exfoliating bark at heights of between seven metres and 24 metres on dead forest trees. All but one have been either in Sydneys Dale or along the western coastal terrace between Anderson's Dale and Martin Point. These trees are highly prone to falling over. Indications are that dead trees with exfoliating bark are very uncommon. This suggests that the preferred roosting habitat is in limited supply and could be a limiting factor to the population's recovery. Sample sizes are small, so it is not possible to rule out a sampling error. Even so, it could well indicate that live trees have become unsuitable as roost sites sometime since the 1980s or 1990s. More research is needed to confirm this.

One possible cause of this might be the explosion in the number of yellow crazy ants, which forage on scale insects in the canopy of live trees. Such a scenario might explain the apparent shift away from live trees as roost sites, which might lead to roost sites becoming a limiting factor and in turn explain the declines in abundance. Nevertheless, the geographical distribution of yellow crazy ants is poorly correlated with the westward contraction of the pipistrelle.

3. Christmas Island flying fox

The Christmas Island flying fox *Pteropus [melanotus] natalis* is endemic to Christmas Island in the Indian Ocean. It is the only frugivorous bat on the island. It is not currently listed as a threatened species under the EPBC Act or by the IUCN (2006). The role of flying foxes in seed dispersal and pollination make them important species in forest ecosystems throughout the African and Asian tropics and especially on oceanic islands (for example, Cox et al. 1991; Elmqvist et al. 1992; Richards 1995).

Flying foxes have complex social and breeding structures. Many species roost communally by day in clusters of trees known as ‘camps’. The locations of camps tend to be permanent, although the numbers of occupants can fluctuate greatly. Camp roosting appears to be an aspect of the life-history of flying foxes that is extremely important to their conservation biology and management (for example, Vardon et al. 2001).

This report reviews available information on the Christmas Island flying fox, its current abundance and distribution, and examines some basic biological factors to determine the significance of the species in the Christmas Island forest ecosystem. Assessment and monitoring of the population were undertaken between November 2005 and December 2006.

3.1 Summary of the results

There is good evidence from this and other studies that numbers of the Christmas Island flying fox have declined. From 1897 to the mid-nineteenth century, it was considered very common all over the island. It was seen feeding in the forest in great numbers and raiding fruit crops in the settlement. By 1984 the population was estimated to be a possible 6,000 individuals in six camps. In 2006, there were fewer than 2,000 individuals in three camps, with three camps appearing to have been permanently abandoned (Table 9).

Table 9: Summary of trends in distribution and abundance of the Christmas Island flying fox from 1984 to 2006

Year	Summary of status, distribution, abundance	Reference
1897	Common	Andrews 1900
1932	Common	Gibson-Hill 1947
1947	Common	Gibson-Hill 1947
1984	Common: population estimated to be about 6,000 individuals	Tidemann 1985
2002	Population estimated to be about 500–1,000 individuals	Corbett et al. 2003
Dec 2005 - Aug 2006	Estimates of a maximum of 450 individuals in camps and a smaller but unknown number roosting away from camps gave a total of less than 1,000 individuals.	This report
Sep 2006	A minimum of 1,381 flying foxes at the three camps, at a time when most of the population congregates at camps	This report

The Christmas Island flying fox is largely diurnal, and so can be easily seen in flight during the day. It continues to feed over most of the island, and there are no baseline data to assess shifts in micro-distribution. The low reporting rates (a measure of relative abundance) of the Christmas Island flying fox in the ‘dog’s head’ area of the north-east (Sector D) from the diurnal and nocturnal forest surveys probably reflect the large amount of clearing that has occurred there as well. The distribution of roosting away from camps is not known.

All recorded campsites have been located on the coastal terrace or around the first inland cliff. This is possibly because coastal winds facilitate easier take-off from roost sites and subsequent gain in altitude. The camps are invariably in semi-deciduous rainforest, although floristic and structural features vary considerably between camps. Four of the six historical camps have been on the east coast with one each on the north and south coasts. Seasonal variation in the occupancy of camps is frequent but not fully understood. It appears that much of the population congregates in the camps in September and October to mate.

Lack of knowledge is a significant issue for the ongoing viability of the Christmas Island flying fox. Specific gaps include:

- causes and exact timing of the decline
- knowledge of many of the potential threats
- population biology—inadequate data exist on basic population parameters such as fecundity, natality, recruitment and mortality rates, effective breeding life-span, and sex ratio
- camp function and structure—identifying and implementing conservation management initiatives will be difficult if not impossible without further understanding of the camps
- ecosystem services—the flying fox is an important pollinator and seed disperser on Christmas Island, but no specific research has been carried out on this topic.

Despite these gaps, there can be no doubt that the Christmas Island flying fox is an important species on Christmas Island because of its role in pollination and seed dispersal for trees, some of which are found only on Christmas Island. The Christmas Island flying fox has been recorded feeding on fruits, flowers and leaves; but the relative importance of different species in the diet is poorly known.

Inadequate knowledge of the timing and pattern of the decline in the Christmas Island flying fox hinders the analysis of reasons for the decline, other than to propose that it is caused by a combination of factors. It is considered unlikely that construction of the IRPC has contributed to the decline.

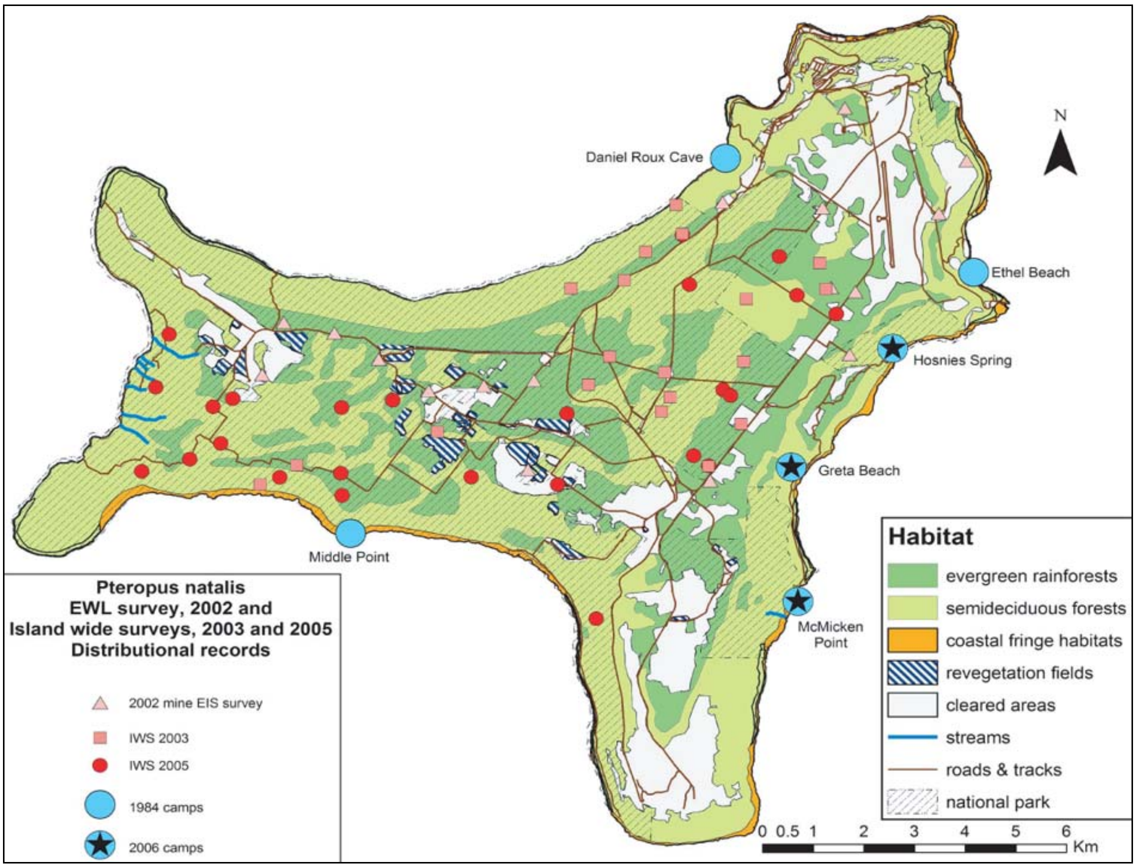
3.2 Distribution and abundance

3.2.1 Historical studies

The Christmas Island flying fox was once very common all over Christmas Island. Lister (1888, p. 516) reported ‘a tree in Flying Fish Cove where they used to hang up during the day’ in 1887. The species was seen feeding in the forest in great numbers, causing much destruction to fruit crops in the settlement (Andrews 1900). Andrews also reported seeing a dead tree near the coast that was covered in hundreds of bats in March, but deserted a week or two later. In 1939–41, the Christmas Island flying fox was ‘flourishing’ and was a nuisance as it fed freely on cultivated papayas (Gibson-Hill 1947). Graham Collins (in Tidemann 1985) reported seeing two or three trees near Daniel Roux Cave, on the northern coast, covered in bats in July to August 1970, but not subsequently.

In 1984, the total population was estimated at about 6,000 individuals. Six communal camps held about 3,500 individuals. This included more than 2,000 Christmas Island flying foxes at Hosnies Spring (Table 10; Figure 15). Three camps were maternity camps that were occupied year-round, and two were seasonally occupied by juvenile males. An additional 2,500 Christmas Island flying foxes roosted singly or in small groups (three to four individuals) scattered across the island. Another camp was discovered at McMicken Point/Dolly Beach a few years later (H. Yorkston and T. Stokes in Tidemann et al. n.d.).

Figure 15: Locations of camps of Christmas Island flying fox and recent sighting data



Count data from various studies (Table 10) show a tendency for camps to be heavily occupied in the dry season and deserted in the wet season. At Ethel Beach, numbers peaked from August to early November and dropped to nil between late November and April. No seasonal pattern is apparent in the Middle Point data. In addition to camps at counts, T. Stokes (Parks Australia archives) also made three counts at what appears to have been a temporary communal roost in two Celtis trees at the golf course in 1985: 12 August (15 bats); 18 September (30 bats); and 1 November (0 bats).

Table 10: Counts of flying fox at six known camps in the mid-1980s

Camp	Date	Ground count	Exit count[1]	Reference
Daniel Roux Cave	March 1984	0		Tidemann 1985
Daniel Roux Cave	21–22 September 1984	150		Tidemann 1985
Daniel Roux Cave	March 1985	0		T. Stokes in Tidemann 1985
Hosnies Spring1	March 1984		150–200	Tidemann 1985
Hosnies Spring1	9 September 1984		1,543	Tidemann 1985
Hosnies Spring1	17 September 1984		2,121	Tidemann 1985
Ethel Beach	13 September 1984		c. 130	Tidemann 1985
Ethel Beach	February 1985	0		T. Stokes[2]
Ethel Beach	31 May 1985	8		T. Stokes[2]
Ethel Beach	5 July 1985	40		T. Stokes[2]
Ethel Beach	2 August 1985	c. 200		T. Stokes[2]
Ethel Beach	30 August 1985	c. 40–50		T. Stokes[2]
Ethel Beach	30 September 1985	c. 300		T. Stokes[2]
Ethel Beach	1 November 1985	c. 350		T. Stokes[2]
Ethel Beach	12 November 1985	0		T. Stokes[2]
Ethel Beach	5 December 1985	0		T. Stokes[2]
Ethel Beach	9 January 1986	0		T. Stokes[2]
Ethel Beach	7 February 1986	0		T. Stokes[2]
Ethel Beach	6 March 1986	0		T. Stokes[2]
Ethel Beach	April 1986	0		T. Stokes[2]
Ethel Beach	12 June 1986	20–30		T. Stokes[2]
Ethel Beach	15 July 1986	c. 200		T. Stokes[2]
Ethel Beach	8 September 1986	c. 130		T. Stokes[2]
Greta Beach	March 1984	300		Tidemann 1985, Tidemann et al. n.d.
Greta Beach	September 1984	>1,000		Tidemann 1985
Middle Point	1 June 1984	c. 200		T. Stokes[2]
Middle Point	5 September 1984	about 300		Tidemann 1985
Middle Point	6 March 1985	c. 50		T. Stokes[2]
Middle Point	17 May 1985	c. 30		T. Stokes[2]
Middle Point	24 July 1985	c. 8		T. Stokes[2]
Middle Point	4 October 1985	c. 12		T. Stokes[2]
Middle Point	18 December 1985	0		T. Stokes[2]
Middle Point	9 May 1986	c. 10		T. Stokes[2]
Dolly Beach	March 1984	0		Tidemann 1985
Dolly Beach	September 1984	0		Tidemann 1985
Dolly Beach (McMicken Point)[3]	November 1988	100		Tidemann et al. n.d.

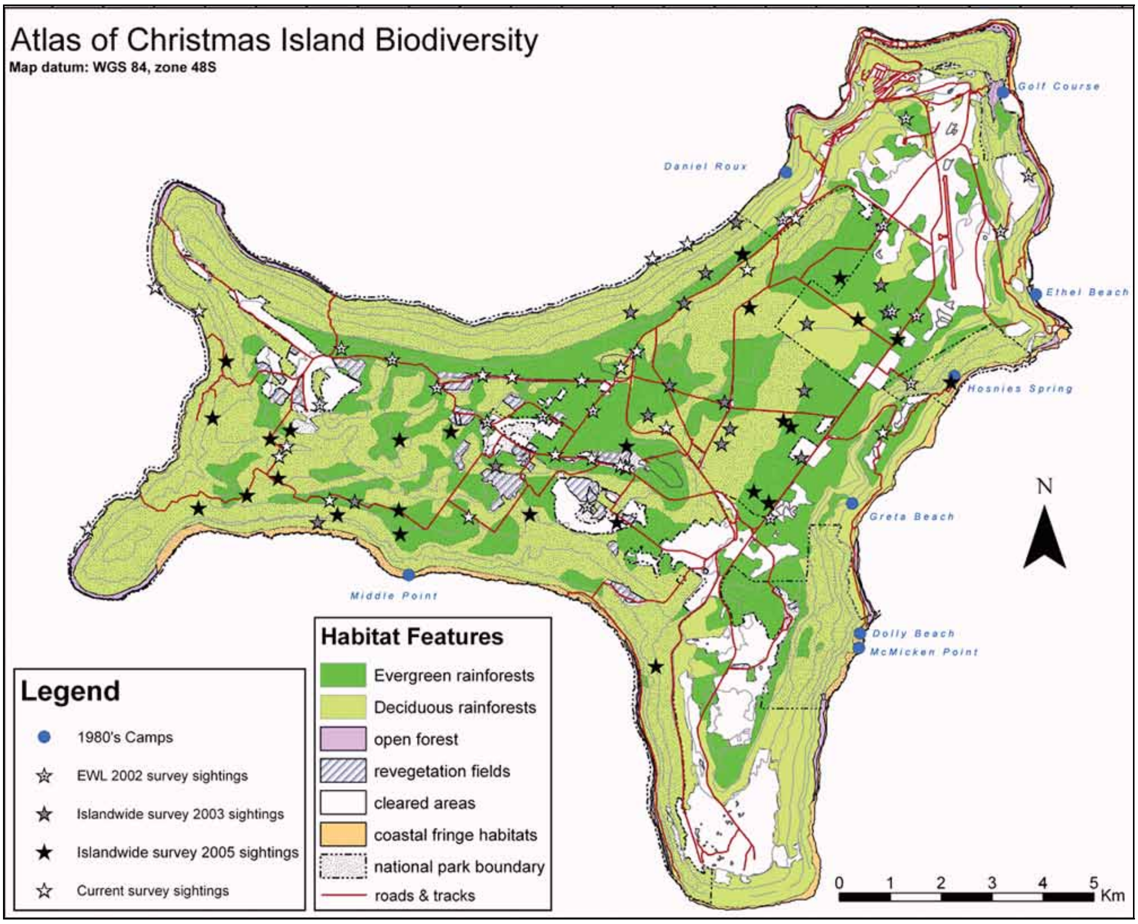
1 = based on exit counts for Hosnies Spring from Margaret Knoll
2 = T Stokes and H. Yorston: unpublished data in ANPWS file notes archived at Parks Australia North, Christmas Island
3 = McMicken Point and Dolly Beach appear to be two names for the same colony on the point south of Dolly Beach (H. Yorkston in lit.)

By the end of the 1990s, anecdotal evidence collected from long-term residents of the island suggested that dramatic population declines of the Christmas Island flying fox had occurred. In 1985 the total population was estimated to be in the order of 6,000 (Tidemann 1985).

As part of an environmental impact assessment in 2002 for proposed new mining leases, Corbett et al. (2003) undertook a brief assessment of the abundance and distribution of the Christmas Island flying fox. They made only 17 records, totalling a maximum of 30 individual Christmas Island flying foxes across the entire island. They did not find any of the historical camps, but it is unclear whether they actually searched for any. They concluded that the population was 'probably in the order of 500–1,000 individuals that are dispersed across the island in small groups and apparently use temporary roosts' (Corbett et al. 2003, p. 48).

Surveys in 2003 and 2005 recorded 20 individual Christmas Island flying foxes and the 2005 island-wide survey recorded 26 individuals from about 1,000 survey points across the entire island, spaced at approximate intervals of 365 metres (Figure 16). In 2003, virtually all records came from the Central Plateau, whereas in 2005 the records were spread more evenly across the island. Neither survey recorded any numbers at South Point, North West Point or the north-eastern part of the island (see O'Dowd and Green 2000).

Figure 16: Christmas Island flying fox sightings from 2002 onwards



3.2.2 Current studies

In 2006, only three of six known camps were occupied, and these collectively contained no more than 500 individuals between December 2005 and August 2006. However, in September 2006, a total of 1,381 individuals was counted. Recent surveys have detected little change in the feeding distribution of the Christmas Island flying fox. It continues to feed over most of the island, but there are no baseline data to assess shifts in micro-distribution.

Colony counts

During seabird surveys in 2002–05, searches for Christmas Island flying fox camps at the Middle Point, Daniel Roux and golf course camps in April and September failed to record any Christmas Island flying foxes. In December 2003, searches for the Hosnies Spring and Greta camps were similarly unsuccessful.

In December 2005, the BMP embarked on a program to locate camps and conduct frequent ground and exit counts. The survey methods are described in Appendix C. Hosnies Spring and McMicken Point (Dolly Beach) camps were located early in the study. Extensive searches for Greta and especially Middle Point camps (three dedicated survey days) were unsuccessful. A camp was located near Greta Beach during yellow crazy ant survey work in June 2006. In addition, searches were made of the Circuit Track and Dales areas, and exit counts were made at Stronach Knoll, Ryan Hill, Field 25 and Douglas Point in an attempt to identify additional camps.

Between December 2005 and August 2006, no more than 500 Christmas Island flying foxes were counted in any one week from all combinations of ground and exit counts (Table 11). In September 2006, there were 1,381 Christmas Island flying foxes, of which 670 exited McMicken Point, 199 exited Hosnies Spring, and 512 camped at Greta Beach. Exit counts for Hosnies Spring were made from the lookout platform at Margaret Knoll, and for McMicken Point from the top of the boardwalk on the headland at the northern side of Dolly Beach.

Some tentative seasonal trends are suggested by the data in Table 11. Firstly, in the middle of 2006, numbers declined steeply at Hosnies Spring and apparently increased simultaneously at Greta. This could indicate some movement between camps. Secondly, most bats deserted all camps during the wet season, as is also suggested in the 1980s data. Thirdly, numbers in the camps increased by 100 per cent or more in September and October. This increase is likely to reflect a seasonal aggregation in the camps for mating. It also provides some support for Tidemann’s (1985) suggestion that about half of the population roosts away from the camps for much of the year.

Northern coastline transects

In 1984, Tidemann (1985) used a boat to record ‘large numbers’ of flying fox roosting singly or in small groups along the northern coastline between Smith Point and West White Beach. He assumed that much of the coastline might be like this. As part of this BMP in November and December 2006, similar counts were made along the coastline between Smith Point and Egeria Point (about twice the distance) but no more than 12 Christmas Island flying foxes were recorded in any transect. Differences in the time of year might have been a factor.

Table 11: Counts of Christmas Island flying fox at three known camps in 2004–06

Camp1	Date	Ground count[2]	Exit count[2]
Hosnies Spring[3]	13 August 2004	>121	
Hosnies Spring	21 September 2005		>32
Hosnies Spring[3]	2 November 2004	299	
Hosnies Spring	5 December 2005		148
Hosnies Spring	12 December 2005		139
Hosnies Spring	14 December 2005		87
Hosnies Spring	16 December 2005	34	
Hosnies Spring	19 December 2005		6
Hosnies Spring	20 December 2005		7
Hosnies Spring	21 December 2005	18	1
Hosnies Spring	26 December 2005		36
Hosnies Spring	2 January 2006		21
Hosnies Spring	5 January 2006	2	6
Hosnies Spring	10 January 2006		29
Hosnies Spring	16 January 2006	122	58
Hosnies Spring	21 January 2006		141
Hosnies Spring	23 January 2006	22	9
Hosnies Spring	6 February 2006		10
Hosnies Spring	21 February 2006	267	
Hosnies Spring	22 February 2006		141
Hosnies Spring	1 March 2006		90
Hosnies Spring	6 March 2006	238	27
Hosnies Spring	27 March 2006	0	
Hosnies Spring	26 April 2006	62	
Hosnies Spring	9 May 2006	287	
Hosnies Spring	30 May 2006		84
Hosnies Spring	6 June 2006	5	
Hosnies Spring	12 June 2006		86
Hosnies Spring	20 June 2006	9	100
Hosnies Spring	18 July 2006	11	80
Hosnies Spring	20 July 2006	0	
Hosnies Spring	21 July 2006		105
Hosnies Spring	23 July 2006	10	54
Hosnies Spring	26 July 2006	15	160
Hosnies Spring	1 August 2006	31	167
Hosnies Spring	5 August 2006		178
Hosnies Spring	9 August 2006	23	114
Hosnies Spring	11 August 2006	32	185
Hosnies Spring	17 August 2006	27	212
Hosnies Spring	21 August 2006	82	193
Greta Beach	16 June 2006	276	
Greta Beach	19 June 2006	310	
Greta Beach	20 June 2006	289	
Greta Beach	2 July 2006	280	
Greta Beach	18 July 2006	207	
Greta Beach	24 July 2006	32	
Greta Beach	26 July 2006	35	
Greta Beach	1 August 2006	14	
Greta Beach	9 August 2006	7	
Greta Beach	11 August 2006	12	
Greta Beach	17 August 2006	5	
Greta Beach	21 August 2006	35	
McMicken Point	22 December 2005	5	
McMicken Point	4 January 2006	12	14
McMicken Point	24 January 2006	13	3
McMicken Point	7 February 2006	83	
McMicken Point	22 February 2006	127	217
McMicken Point	9 March 2006	224	146
McMicken Point	29 March 2006	138	163
McMicken Point	20 June 2006	37	180
McMicken Point	26 July 2006		221
McMicken Point	2 August 2006	79	168
McMicken Point	11 August 2006	23	92
McMicken Point	17 August 2006	68	106

[1] 'McMicken Point' and 'Dolly Beach' camps assumed to be the same

[2] Exit counts for Hosnies Spring made from Margaret Knoll and for McMicken Point from the boardwalk at the cliff top on the north side of Dolly Beach

[3] Source for Hosnies Spring on 13 August and 2 November: M Jeffries pers. comm. All other data from this BMP

Forest surveys for Christmas Island flying foxes

The 2005–06 forest bird survey recorded diurnal sightings of the Christmas Island flying fox. The results are shown in Figure 17; and Table 12 shows the reporting rates by habitat. A total of 511 counts was made (one count was missed at one site). Christmas Island flying foxes were recorded in 24 counts (a total reporting rate of 4.7 per cent) at 22 (17.2 per cent) of the 128 sites. The forest bird survey methods are described in Appendix B.

In June–July 2006, the survey was repeated at night to specifically target nocturnally active Christmas Island flying foxes. The results show the reporting rates by habitat (Table 13). A total of 436 (= 4 x 109) surveys was made. Christmas Island flying foxes were recorded in 82 counts (a total reporting rate of 18.8 per cent) at 45 (41.3 per cent) of the 109 sites.

Sightings of Christmas Island flying foxes were significantly fewer in sector D (the north-east) than in sector E (the central east) of the island ($\chi^2 = 20.650$, df = 5, P = 0.0009). There were no significant differences in reporting rates between habitats ($\chi^2 = 3.247$, df = 3, P = 0.3552), time of day ($\chi^2 = 6.0786$, df = 6, P = 0.4145) or between the four replicates surveys ($\chi^2 = 1.000$, df = 3, P = 0.8013). The latter demonstrates good consistency of variance (homoscedacity) within the four surveys, which suggests that sampling bias was at least consistent between surveys. Seasonality could not be assessed because the survey spanned only two months.

Figure 17: Christmas Island flying fox sightings from the 2005 forest bird survey

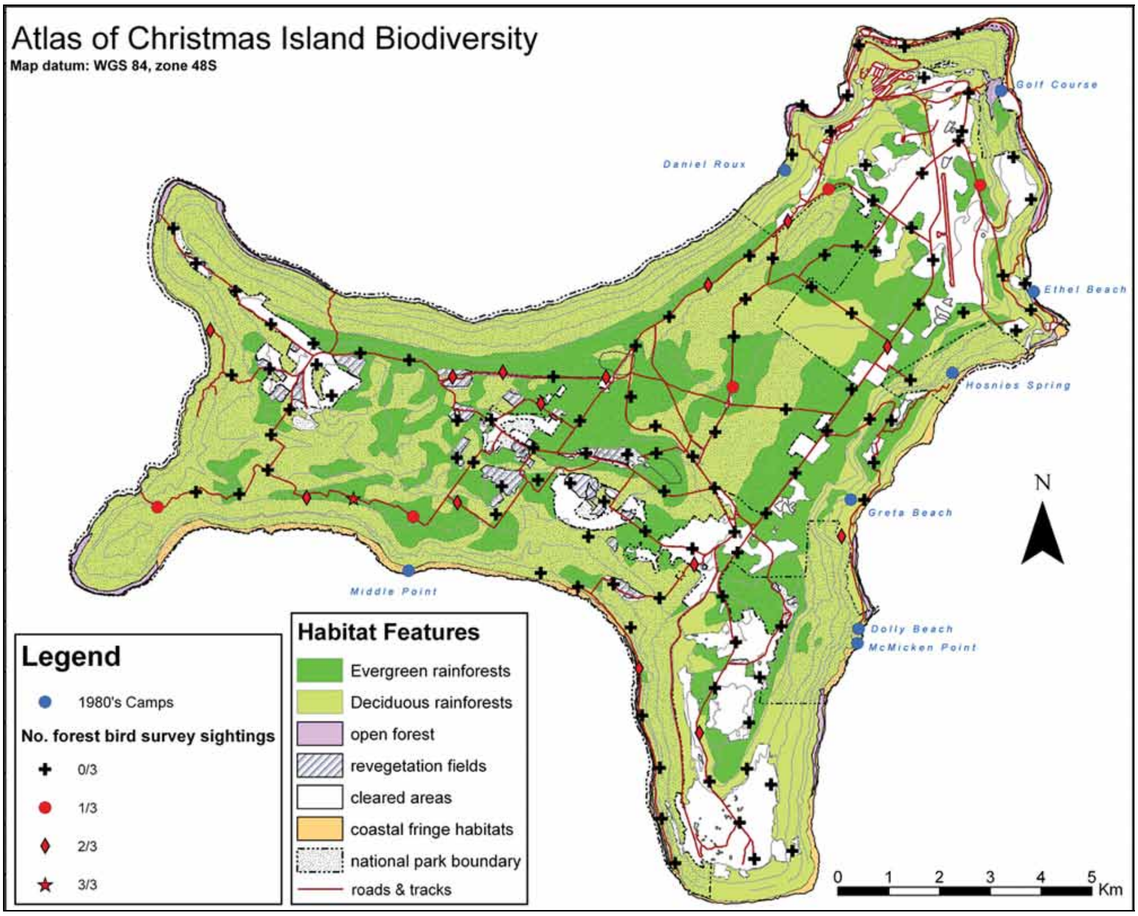


Table 12: Reporting rates by habitat for Christmas Island flying fox from the diurnal forest survey, 2005–06

Habitat type[1]	**A**	**B**	**C**	**D**	**All**
No. sites in each habitat	33	26	61	8	128
No. times *P. melanotus* recorded	10	7	5	2	24
Minimum reporting rate[2]	0%	0%	0%	0%	0%
Maximum reporting rate	50%	25%	50%	25%	50%
Overall reporting rate[3]	7.58%	6.73%	2.06%	6.25%	4.70%

[1] Habitat definitions: A = primary evergreen rainforest; B = primary semi-deciduous rainforest; C = disturbed habitats; D = edge habitats including either A and C or B and C

[2] Minimum and maximum reporting rates are calculated from four replicate surveys within sites (i.e. they can only be 0, 25, 50, 75 or 100%)

[3] Overall reporting rates are calculated from the total 128 x 4 – 1 = 511 surveys

Table 13: Reporting rates by habitat for Christmas Island flying fox from the nocturnal forest survey, 2006

Habitat type[1]	**A**	**B**	**C**	**D**	**All**
No. sites in each habitat	29	26	47	7	109
No. times *P. melanotus* recorded	22	23	35	2	82
Minimum reporting rate[2]	0	0	0	0	0
Maximum reporting rate	75%	75%	75%	25%	75%
Overall reporting rate[3]	18.97%	22.12%	18.62%	7.14%	18.81%

(See Table 12 for explanations)

3.2.3 Trends in abundance

Between 1984 and 2002, the Christmas Island flying fox population experienced a severe decline in abundance, but the magnitude of the decline is not known precisely. The most conservative estimate is a 67 per cent decline based on 6,000 bats in 1984 (Tidemann 1985) versus 2,000 bats in 2006 (this report). A higher estimate of 85 per cent could be based on 10,000 bats in 1984 (Tidemann in Mickelburgh et al. 1992; Tidemann in Duncan et al. 1999) versus 1,500 in 2006 (this report). The most parsimonious estimate would be a 75 per cent decline from 6,000 in 1984 to 1,500 in 2006 (this report).

The first report of the decline was from surveys in August 2002 (Corbett et al. 2003). Corbett et al. argued that the decline occurred suddenly on 27 March 1988 due to a storm. Conversely, two independent lines of evidence could indicate that the decline began after 1988. Firstly, Tidemann et al. (n.d.) contained field data from after March 1988 (for example, ground counts of the McMicken Point camp in November 1988), but did not mention any decline. However, the number of fallen trees during Tidemann’s 1988 visit and the focus on rats and cats at that time meant that there were no thorough surveys of the Christmas Island flying fox (C. Tidemann in lit.). Secondly, anecdotal evidence compiled as part of this BMP suggests that a sudden decline occurred in the mid-1990s (Walker/Orchard, pers. comm.). The scientific data show only that the Christmas Island flying fox population declined by about 83–95 per cent between September 1984 and August 2002 (Tidemann 1985; Corbett et al. 2003; Dale in prep.; this report).

It remains unclear whether there was:

- one crash from which the population has not recovered
- a gradual decline, or
- a stepped decline (that is, a gradual decline initiated and/or punctuated by a crash or crashes).

It is also plausible that some decline occurred before 1984. Hunting by humans likely occurred from soon after settlement (in 1895) up to and beyond 1984 (Tidemann 1985), although Tidemann found no evidence of any impacts to population levels from hunting.

3.3 Biology

3.3.1 Population and breeding biology

There is very little sexual dimorphism in size or outward appearance in the Christmas Island flying fox. The longevity is not known, but males apparently have a lower life expectancy than females. The rates of natality, mortality and recruitment are not known. Breeding is seasonal and annual. The peak period of birthing appears to be in the wet season, from December to February, although with some spread.

The precise structure and function of existing camps are not known, but they appear to be closely tied to social structure and reproductive strategies. The Christmas Island flying fox has a polygamous mating system with a small proportion of the male population monopolising mating opportunities with most adult females.

Population structure

Most of the information on population structure comes from Tidemann (1985), and was based on the dissection of about 115 specimens. The primary sex ratio is close to 1:1 (55 males: 60 females in Tidemann’s collection). The single population is closed with no evidence of immigration or emigration. The longevity is not known, but males apparently have a lower life expectancy than females (Tidemann 1985). The rates of natality, mortality and recruitment are not known. Tidemann (1985) estimated that sexual maturity is reached by females at about six months of age and by males at about 18 months of age. Consequently (in 1984) the age structure of the population was highly skewed, containing:

- more mature females than males
- more mature females than immature females
- more immature males than mature males.

This disparity would have produced an operational sex ratio of about 3:1 females to males (C. Tidemann in lit.). Females also become volant at an earlier age than males. Females rarely (or never) produce twins (Tidemann 1985), so the natality rate is less than one pup per mature female per year. Taking juvenile mortality into account, individuals must breed for at least several years on average to replace the population. Gestation lasts about five months (Tidemann 1985). Martin and McIlwee (2001) argued that all species of flying foxes have an average generation time of between five and nine years, with the lower estimates found in stressed or declining populations. Their data were from captive animals. C. Tidemann (in lit.) considers the generation length to be about four years in *Pteropus poliocephalus* in the wild, and therefore closer to two years in the Christmas Island flying fox. An accurate figure is not available.

Birthing period

The peak period of birthing appears to be in the wet season, from December to February. Andrews (1900) recorded full-term foetuses in late December (see also Thomas 1887). In a sample of 42 females collected in September 1984, there were three with small foetuses and one that had recently given birth (Tidemann 1985). In January, three females had large foetuses, one had recently given birth, and a large suckling pup was found and handed to the then Australian National Parks and Wildlife Service. In a sample of 42 females collected in March 1984, none was pregnant but many were lactating. From these data, Tidemann (1985) estimated that the peak birthing period was in February. Nevertheless, there is considerable spread and there could be variation between years.

In 2005–06, females carrying young were seen from Margaret Knoll on 14, 20, 21 and 26 December, and 2, 5, 16 and 23 January, but not subsequently. These observations suggest that either:

- the period of peak birthing was earlier in 2006 than in 1984–85, or
- Tidemann slightly overestimated the date of the peak in birthing and therefore probably the length of the gestation period as well.

Females of all *Pteropus* species carry their single suckling pup (which clings to its mother's belly fur) between roost sites and foraging areas for some time after birth. As the pups get older and larger they are left at roost sites (including camps), sometimes in crèches, until they are volant. No specific information on this process is available for the Christmas Island flying fox.

Mating system

The mating systems of *Pteropus* species are exceedingly complex. Tidemann (1985) suggested that Christmas Island flying fox on the island might use a system where a small proportion of dominant mature males inseminate the large majority of females in camps. Thus, a handful of males sire all the pups and most sire none. More research is needed to confirm this.

4.3.2 Behaviour

Diel patterns of activity

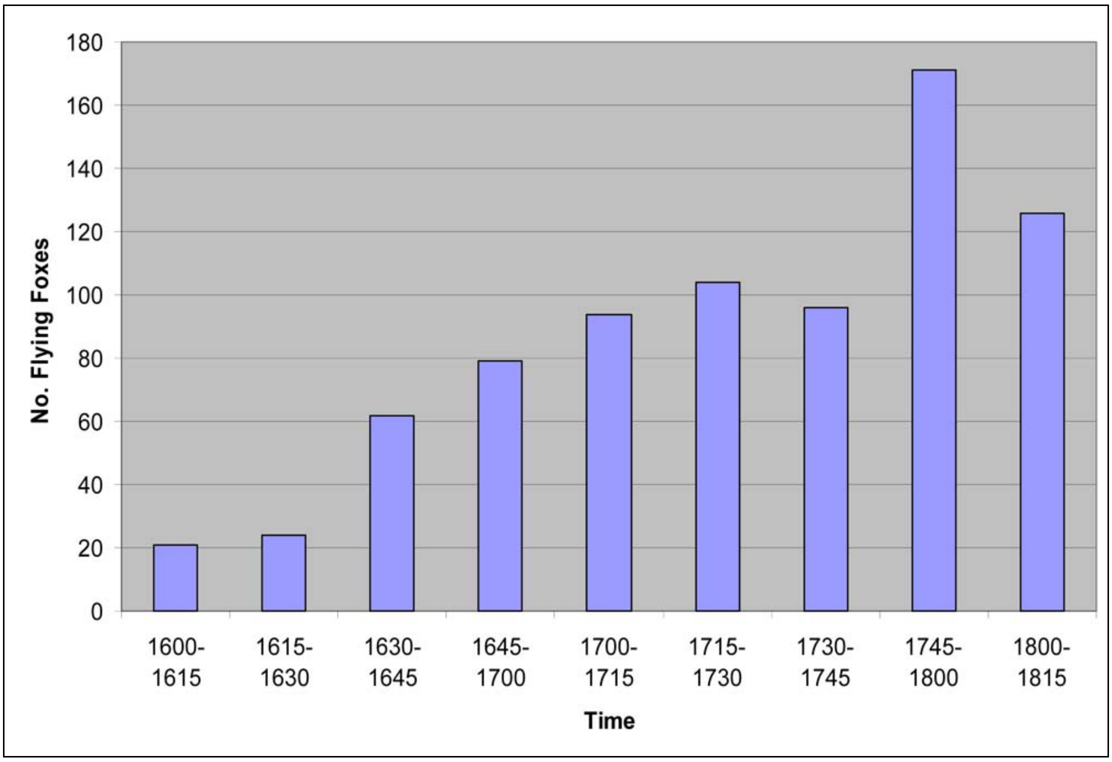
Andrews noted that the Christmas Island flying fox:

> has to a very great extent abandoned the nocturnal habits usual in the group. Several might often be seen sailing and circling high in the air in the hot sunlight, sometimes even at midday, and I have also frequently seen them feeding in the daytime. (Andrews 1900, p. 25)

Tidemann (1985) found that most foraging occurred during daylight, with peaks from 2:30 pm to 6:30 pm and at about 6 am. Reviewing diurnal activity in *Pteropus* spp., Tidemann suggested that it occurred regularly in island populations where native predators were absent, and that this was unaffected in the Christmas Island flying fox even after 80 years of human predation.

Flying fox activity (both exits from camps and arrivals at foraging trees) rarely began before 4 pm and peaked at about dusk (5:45 pm to 6:15 pm at that time). This is shown in Figure 18.

Figure 18: Christmas Island flying fox flying activity from this survey



Note: while all dusk vantage point counts went at least until 6 pm, not all continued to 6:15 pm.

The data show that the diurnal habit of the flying fox has diminished since the 1890s and the 1980s. During constant fieldwork on the island between 2002 and 2006, flying foxes were rarely seen feeding before the fading light prior to dusk. It is not clear whether this is due to a change in habit, a smaller flying fox population available to observe, or less need to be diurnal with fewer flying foxes competing for resources.

A similar pattern of diminished diurnal behaviour has been identified in the island’s other endemic bat species: the insectivorous Christmas Island pipistrelle.

Foraging

According to Tidemann (1985), flying foxes appeared to space randomly with respect to each other when foraging. However, more than six individuals were rarely recorded in one tree, suggesting some degree of spacing. He could not determine whether they foraged in flocks or individually. Feeding height varied with canopy height and food availability. He observed sudden changes in diet at a population level. In March 1984, limited radio tracking suggested that foraging ranges might be small, with commuting distances of about 1.5 kilometres between camps and food sources. In September

1984, observations of exiting bats suggested that commuting distances were frequently more than five kilometres. This might suggest that maternity camps are located near the best feeding trees during the pupping and weaning periods (the wet season), to minimise foraging distances for females that are carrying young or leaving their young in crèches.

There are no recent data on foraging ranges. However, the localised distribution of camps and their periodic abandonment, compared with the island-wide distribution of the bats, strongly suggest that much of the population roams all over the island.

Camp structure and function

'Pteropodids are well known for their habit of communal roosting' and 'once formed, a pteropodid camp is likely to be used for a long time, despite interference from man or other agents such as cyclones' (Tidemann 1985, pp. 45–6). Although the mechanisms of camp formation are not known, they appear to be closely tied to social structure and reproductive strategies, which differ somewhat between species. The Christmas Island flying fox has a polygamous mating system with a small proportion of the male population monopolising all adult females (Tidemann 1985). This led Tidemann (1985) to the tentative conclusion that the Christmas Island flying fox has a social system similar to *Pteropus giganteus*, in which adults of both sexes live year-round in camps at traditional sites. However, he also noted that, unlike *P. giganteus*, many individual flying foxes roost singly and in small groups well away from camps. Additionally, count data from the 1980s (Table 10) and 2002–06 (Table 11) indicate that camp numbers are not stable but follow seasonal cycles on Christmas Island.

In some instances, different camps of a single *Pteropus* species may have different social structures and functions. Many species segregate into unisexual camps between copulation and parturition with missed flocks during lactation (Tidemann 1985). Rank orders exist between individuals, particularly males. Tidemann estimated the social structure and function of camps of Christmas Island flying fox based on the age and sex classes of individuals collected from them (Table 14). All of the data are from September 1984, so there is little information on changes in structure and function through the year.

Table 14: Summary of assumed camp structure and function for Christmas Island flying fox

Camp	Structure	Function
Hosnies Spring	10 bats collected: 9 pregnant females, 1 mature male	Maternity camp
Middle Point	10 bats collected: 9 pregnant females, 1 mature male	Maternity camp
Greta Beach	13 bats collected: 6 pregnant females, 4 mature males, 1 juvenile female, 4 juvenile males	Mixed maternity and juvenile camp?
Ethel Beach	10 bats collected: 2 juvenile females, 8 juvenile males	Juvenile camp
Daniel Roux Cave	10 bats collected: 10 males	All-male camp
McMicken Point	None	Unknown

Based on collections made in 1984 (Tidemann 1985)

The disappearance of the Ethel Beach camp since the 1980s could well reflect a decline in the rate of juvenile survival and consequently the rate of recruitment in the population. Alternatively, it could reflect the loss of a specific feeding resource, for example, resulting from construction of the casino resort in the early 1990s. The disappearance of the Middle Point camp could reflect a reduction in the breeding population.

The disappearance of the Daniel Roux Cave camp may reflect a population issue, direct displacement by dust fall-out from the phosphate dryers located on the plateau above, or indirect displacement by dust fall-out smothering local food resources. Dust fall-out has been implicated in the abandonment of the Christmas Island frigatebird nesting colony below the dryers between 1971 and 2003 (Stokes 1988).

3.3.3 Habitat

Location of camps and roost sites

All recorded campsites have been located on the coastal terrace or around the first inland cliff (Tidemann 1985; Tidemann et al. n.d.; see also Figure 16). They are invariably in semi-deciduous rainforest, although floristic and structural features vary considerably between camps. Four of the six historical camps have been on the east coast with one each on the north and south coasts.

Seasonal variation in the occupancy of camps is common among *Pteropus* species, including the Christmas Island flying fox (Tidemann 1985; Tables 1 and 2). Seasonal shifts of camps in response to shifting food resources could be less important on Christmas Island, being only 135 square kilometres, than elsewhere. Seasonal dispersal from camps to avoid competition for food resources could explain both the fluctuations in camp numbers and the use of small roosts away from camps. Seasonal use of denser vegetation for protection from rain and seasons of lower humidity could also explain these patterns. There is no evidence that predators influenced the location of traditional camps of the Christmas Island flying fox.

The camps of Christmas Island flying fox are always near the coast (Tidemann 1985) and usually below the first inland cliff (the exact location of the Middle Point camp is not known, but it might have been on the slopes of the inland cliff near the access track to the lower terrace (H. Yorkston in lit.)). Tidemann (1985) suggested that this is likely to be because coastal winds facilitate easier take-off from roost sites and subsequent gain in altitude. The south-east trade winds in the dry season push against the first inland cliff of the east coast, causing a rising current, and Christmas Island flying foxes can be seen using this current when they exit Hosnies Spring camp. Conversely, we have observed that, on windless afternoons, they struggle to clear the inland cliff; they frequently land in vegetation several times on the way to the cliff top. There was a significant relationship between wind speed and exit counts at Margaret Knoll on 13 different days between May and August 2006 (Pearson's correlation: $r = 0.70392$; $df = 12$; $P = <0.01$).

The seasonal variations of the trade winds could explain the periodic desertion of camps. However, wind patterns are unlikely to determine the location of camps on the coastal terraces. The majority of foraging habitat is on the plateau (assuming that food resources are more or less evenly spread through the forest), so theoretically, if camps were located on the plateau, then the Christmas Island flying fox would have easier access to plateau resources on windless days and also have easy access to the coastal terraces during windy conditions. In turn, this might suggest that there are more or better food resources on the coastal terraces than elsewhere, at least during the breeding season.

In contrast to the long-term stability of camp locations, individuals and small groups roosting away from camps apparently do not remain consistently in any one tree or vicinity for any length of time. Andrews (1900) notes around December: 'At this time a dead tree near the coast was seen covered with hundreds of these bats, but a week or two afterwards they had completely forsaken it.'

Roosting trees

The Christmas Island flying fox has been recorded using 13 species of tree for roosting (Tidemann 1985; and this BMP): *Barringtonia racemosa, Bruguiera gymnorhiza, Celtis timorensis, Ficus microcarpa, Guettarda speciosa, Gyrocarpus americanus, Hibiscus tiliaceous, Kleinhovia hospita, Ochrosia ackeringae, Macaranga tanarius, Pisonia grandis, Syzygium nervosum and Terminalia catappa*. This list is unlikely to be exhaustive; the flying fox will also roost in vine tangles and possibly in cultivated fruit trees.

Tidemann (1985) noted that roost trees in camps and elsewhere were frequently canopy emergents, suggesting that this was because greater height gives better exposure to winds that facilitate easier take-off. This is plausible, but flying foxes will also roost in stunted trees or low down in canopy emergents (Tidemann 1985; BMP unpubl. data). In the latter situations, they tend to climb to the top of roost trees to exit camps (Dale in prep.).

Being highly mobile, the flying fox utilises most of the vegetated terrestrial habitats present on Christmas Island, to a greater or lesser extent. The following habitat descriptions are based mainly on descriptions from the Director of National Parks (2002).

3.3.4 Habitat descriptions

Semi-deciduous rainforest on lower terraces

Open, semi-deciduous rainforest is typical on the coastal terraces, with scrambling and spiny shrubs and vines. Both vine and canopy forests also occur. Christmas Island flying foxes use this habitat for both feeding and roosting away from camps; and all of the current camps are located in this habitat. This habitat generally has shallow soils prone to dehydration in the dry season. Typical plant species are *Acronychia trifoliolata, Berrya cordifolia, Calophyllum inophyllum, Erythrina variegata, Hibiscus tiliaceus, Kleinhovia hospita, Ochrosia ackeringae, Pandanus elatus, Pisonia grandis, Gyrocarpus americanus* and *Terminalia catappa*.

Semi-deciduous rainforest on shallow soils of higher terraces

Christmas Island flying foxes use this habitat for both feeding and roosting away from camps. This habitat generally has thin soils and exposed limestone pinnacles. Typical canopy species include *Celtis timorensis, Dysoxylum gaudichaudianum, Ficus microcarpa, Arenga listeri, Planchonella nitida* and *Tristiropsis acutangula*. Vegetation has a lower canopy height and is floristically richer than the evergreen rainforest of the plateau. Pockets of deeper soil occur in this region, supporting evergreen rainforest.

Limestone scree slopes and pinnacles

Christmas Island flying foxes use this habitat for both feeding and roosting away from camps. The inland cliffs rise out of the semi-deciduous terrace rainforest and support *Ficus microcarpa, Maclura cochinchinensis, Gyrocarpus americanus, Erythrina variegata, Derringia amaranthoides* and *Dendrocnide* species.

Evergreen rainforest on deeper plateau and terrace soils

Christmas Island flying foxes use this habitat for both feeding and roosting away from camps. This habitat typically has a tall evergreen rainforest with emergent trees to 45 metres. Typical emergent species are *Syzygium nervosum*, *Ficus microcarpa, Planchonella nitida* and *Hernandia ovigera*. The upper canopy includes *Barringtonia racemosa, Inocarpus fagifer, Cryptocarya nitens, Dysoxylum gaudichaudianum* and *Tristiropsis acutangula*. The understorey includes *Arenga listeri, Pandanus elatus, Leea angulata, Ochrosia ackeringae, Pisonia umbellifera* and various shrubs and ferns.

Mangrove forest

There are no coastal mangroves on Christmas Island, but a stand of normally estuarine *Bruguiera gymnorhiza* and *B. sexangula* occurs on a freshwater stream at Hosnies Spring (listed as a Wetland of International Importance under the Ramsar Convention) about 50 metres above sea level. The Christmas Island flying fox camp at Hosnies Spring is located in and adjacent to this mangrove stand (Tidemann 1985). During 2006, no more than 10 per cent of the camp occupants used Bruguiera trees to roost.

Perennially wet areas

Christmas Island flying foxes use this habitat for both feeding and roosting away from camps. Located along springs and seepages, this habitat typically supports *Inocarpus fagifer* and other locally common rainforest tree species. The elevated soil moisture leads to higher transpiration rates and the dense canopy traps the resulting humidity.

Mining fields

Christmas Island flying foxes use this habitat for feeding, although it is apparently neither essential nor important. This habitat is typically limestone pinnacles, boulders, chalk and very thin soils left after mining. These thin soils support the ferns *Nephrolepis multiflora* and *Psilotum nudum* and the exotics *Mimosa invisa, M. pudica* and *Muntingia calabura*. Stockpiles of topsoil are colonised by *Claoxylon indicum*, *Macaranga tanarius*, *Melochia umbellata* and the exotics *Leucaena leucocephala* and *Muntingia calabura*.

Urban and urban-fringe areas

Christmas Island flying foxes use this habitat for feeding, although it is apparently neither essential nor important. Food plants in urban areas are mostly exotic species, including coconut *Cocos nuciferer* and mango *Mangifera odorata* and *M. indica*.

3.3.5 Diet

The Christmas Island flying fox has been recorded to feed on fruits, flowers and, in one case, leaves, from 35 species of vascular plants. Table 15 summarises this information with the plant taxonomy standardised to conform with the Flora of Australia (1993). The list is not likely to be exhaustive. Many of these species will flower and fruit on a prolific and continuous basis, while others flower or fruit in a strongly seasonal pattern. The relative importance of different species in the diet is poorly known.

Native species

Eleven of the native species listed in Table 15 are common and widespread components of the forest canopy on Christmas Island (*Barringtonia racemosa*, *Celtis timorensis, Dysoxylum* sp., *Ficus macrocarpa, Inocarpus* sp., *Macaranga* sp., *Maclura* sp., *Planchonella* sp., *Syzigium nervosum, Terminalia catappa*, *Tristiropsis* sp.); this subset is likely to include the most important foods for the population. K. Walker (pers. comm.) reports that *F. macrocarpa* is not eaten often.

Introduced species

Eighteen of the 35 species recorded in the diet have been introduced to Christmas Island since 1888 (Flora of Australia 1993). Several of these occur only in cultivation and are therefore rare and geographically restricted. Several authors have emphasised the importance of the widespread and abundant weed *Muntingia calabura* (Tidemann 1985, 1987; Tidemann et al. n.d.; Corbett et al. 2003). *M. calabura* fruit was the most frequent component of the Christmas Island flying fox diet in March 1984. In August and early September, *S. nervosum* was the most frequent component; but as sources declined, the Christmas Island flying fox returned to feeding on *M. calabura* in late September (Tidemann 1985; C. Tidemann in lit.). M. Orchard (pers. comm.) noted a decline in the abundance of *M. calabura* along roadsides and a concomitant decline in the number of Christmas Island flying foxes feeding on roadsides between 1992 and 2006. During this BMP, it was observed that coconut and mango are heavily exploited. *Melia* sp., *Psidium* sp. and *Carica* sp. are widespread in disturbed areas, secondary forests and semi-deciduous forests. P. Menkhorst (pers. comm.) saw Christmas Island flying foxes eating large leaves of an unidentified tree (not *Ficus* sp.) on the central plateau in March 2007.

Cultivated fruits

Ten of the 35 species listed in Table 15 are cultivated fruits or ornamental plants. K. Walker has maintained a small tropical fruit orchard at Grants Well on the central plateau since the 1970s. Of the fruits grown there, soursop *Annona muricate* and mango are most preferred by the Christmas Island flying fox. Cashew *Anacardium occidentale* was formerly popular but is no longer grown. Avocado *Persea americana*, black sapote *Diospyros digyna* and sapodilla *Manilkara zapota* are eaten but with less preference than soursop. Christmas Island flying foxes do not seem to eat white sapote *Casimiroa edulis*, carambola *Averrhoa carambola,* jaboticaba *Myrciaria cauliflora* or citrus at Grants Well, despite seasonal availability. They have not been noted eating bananas, although few are available. H'ng Kim Chey (pers. comm.), who has cultivated the plantation on Murray Road for many years, reports that the flying fox feeds on mango and custard apple (*Annona reticulata*) but not banana or avocado.

Jackfruit *Atrocarpus heterophyllus* was formerly used by hunters as bait, wired to trees, to lure flying foxes close to the ground (G. Foo pers. comm.). Custard apple is eaten at the plantation on Murray Road (D. James pers. obs.). The fruit of the umbrella tree *Schefflera actinophylla* was formerly popular but this plant has been extensively culled (M. Orchard pers. comm.). Christmas Island flying foxes have been seen apparently foraging in pomelo trees (*Citrus maxima*) on rare occasions (M. Orchard pers. comm.), where they might have been eating flowers. Banack (1998) recorded orange (*Citrus sinensis*), which is cultivated on Christmas Island, in the diet of Samoan *Pteropus* spp. Water apple or wax jambu *Syzygium aqueum* is also cultivated on Christmas Island and may be eaten. Both native and cultivated populations of hog plum *Spondias cythera* exist on the island (Flora of Australia 1993), although there are only about six individuals (J. Clausen, pers. comm.).

Nelson et al. (2000) found that exotic fruits provided lower levels and less variety of nutrients and minerals for *Pteropus samoensis* in the Samoan archipelago. No comparable information is available for Christmas Island.

Table 15: Summary of dietary information for Christmas Island flying fox

Species	Common name	Food source	Reference*
Anacardium occidentale#	Cashew		8
Annona muricate#	Soursop	Fruit	5,6,8
Annona reticulata#	Custard apple	Fruit	6
Arenga listeri	Arenga palm	Flowers	2,3,5
Atrocarpus heterophyllus#	Jackfruit	Fruit	9
*Barringtonia asiatica**	Box mangrove	Flowers	2,6
Barringtonia racemosa		Flowers	2,3,5
*Carica papaya**	Papaya	Fruit	1,2,3,5,6,7
Celtis timorensis	Stinkwood	Fruit	2,3,5,6
*Cocos nucifera**	Coconut palm	Flowers	2,3,5,6,7

Species	**Common name**	**Food source**	**Reference***
*Citrus maxima**	Pomelo	Fruit	7
Dendrocnide sp.	Stinging tree	Flowers	2,3,5
Diospyros digyna	Black sapote		8
Dysoxylum gaudichaudianum		Flowers	2,3,5
Ficus macrocarpa	Strangler fig	Fruit and leaves?	2,3,5
Inocarpus fagifer	Indian chestnut	Fruit	1,5
Macaranga tanarius		Flowers	2,3,5,6
Maclura cochinchinensis		Fruit	6
Mangifera indica#	Mango	Fruit	2,5,6,7,8
*Mangifera odorata#**	Mango	Fruit	3,5,6,7,8
*Manihot glaziovii**	Ceara rubber tree	Flowers	6
Manilkara zapota#	Sapodilla	Flowers	5,6,8
*Melia azedarach**	White cedar	Flowers	2,3,5
Morinda citrifolia	Cheese fruit	Flowers	2,3,5
*Muntingia calabura**	Cherry	Flowers	2,3,4,5,6,7
Musa spp.#	Banana	Flowers and fruit	1
Persea Americana#	Avocado		8
Physalis sp.		Flowers	2,3,5
Pipturus argentus		Flowers	2,3,5
Planchonella nitida		Flowers and fruit	1,2,3,5
*Psidium guajava**	Guava	Fruit	2,3,5,6
Schefflera actinophylla#	Umbrella tree	Flowers and fruit?	7
Syzigium nervosum		Flowers and fruit	2,3,5
Terminalia catappa	Indian almond	Flowers and fruit	2,3,5,6
Tristiropsis acutangula		Flowers and fruit	2,3,5,6

References: 1 = Andrews (1900);
2 = Tidemann (1985) followed by Mickelburgh et al. (1992);
3 = Tidemann et al. (n.d.);
4 = Corbett et al. (2003);
5 = results from this BMP (2006: includes records from literature);
6 = D. James (unpubl. data, 2004–06);
7 = M. Orchard (pers. comm.);
8 = K. Walker (pers. comm.);
9 = G. Foo (pers. comm.)
* = naturalised species and widespread
# = cultivated species with limited abundance

3.4 Ecosystem services provided by flying fox

Because of their roles as seed dispersers and pollinators, flying foxes are an important element of ecosystems (for example, Marshal 1983; Richards 1990, 1995; Cox et. al. 1991; Fujita and Tuttle 1991; Hodgkison et al. 2003; Nyhagen et al. 2005), especially on islands (Banack 1998; Whittaker 1998). Such species are termed keystone species because of their important roles in shaping and maintaining ecosystems.

3.4.1 Pollination

Flying foxes are extremely important pollinators of many plants, especially rainforest trees, and especially on isolated islands (Marshall 1983; Cox et al. 1991; Whittaker 1998; Cox and Elmqvist 2000). Flying foxes are assumed to play an important role in pollination on Christmas Island (Flora of Australia 1993), although no specific research has been conducted on this subject. Marshall (1983) discussed the importance of pteropodid bats as pollinators and the evolution of floristic characteristics of Chiropterophilous (bat-dependent) plants. Figure 19 lists some forest trees of Christmas Island, which, based on floral characteristics, are likely to be pollinated principally by the Christmas Island flying fox. One of these plant species is endemic to Christmas Island and four occur nowhere else in Australia. Therefore the conservation of these plants within the whole of their Australian range might be dependent on the conservation of the flying fox.

Figure 19: Chiropterophilous plants on Christmas Island probably pollinated primarily by flying foxes

Chiropterophilous plants[1]	**Reference**	**Flowers known from diet?**[2]
Arenga listeri ##	Marshall 1983	Yes
Barringtonia racemosa	Marshall 1983	Yes
Celtis timorensis	Corbett et al. 2003	No
Cocus nucifera	Marshall 1983	Yes
Dysoxylem gaudichaudianum	Corbett et al. 2003	Yes
Hernandia ovigera #	Corbett et al. 2003	No
Inocarpus fagifer #	Corbett et al. 2003	No
Planchonella nitida #	Corbett et al. 2003	Yes
Syzigium nervosum	Corbett et al. 2003	Yes
Terminalia catappa	Corbett et al. 2003	Yes
Tristiropsis acutangula #	Corbett et al. 2003	Yes

[1] See Table 15 for English names; [2] See Table 15 for sources of information on known diet;
Species that occur only on Christmas Island within Australian jurisdiction; ## Species endemic to Christmas Island

3.4.2 Seed dispersal

Flying foxes are extremely important dispersal agents of plant seeds, especially those of rainforest trees, and especially on isolated islands (Marshall 1983; Cox et al. 1991; Banack 1998; Whittaker 1998; Hodgkison et al. 2003; Nyhagen et al. 2005); the Christmas Island flying fox is considered to play an important role in the dispersal of seeds on Christmas Island (Flora of Australia 1993). It is likely that they are important dispersers of several native species, including *Celtis timorensis, Ficus macrocarpa, Inocarpus fagifer, Planchonella nitida, Syzigium nervosum* and *Terminalia catappa*. Tidemann (1985, 1987) considered that the flying fox was a significant disperser of the seeds of the introduced weed *Muntingia calabura*. It is likely that they also disperse the seeds of other widespread weeds, including *Carica papaya, Melia azedarach* and *Psidium guajava*. However, no data exist on this subject. The Christmas Island imperial pigeon *Ducula whartoni* also disperses some of the smaller-seeded species (*Celtis timorensis, Ficus macrocarpa, Syzigium nervosum, Tristiropsis, Muntingia* sp. *and Melia* sp.; BMP unpubl. data). Other species of flying foxes are known to be important dispersal agents of pandanus in Southeast Asia (Fujita and Tuttle 1991), although the two endemic species of pandanus on Christmas Island have not been reported in the diet of its flying fox.

Seeds carried by flying foxes are especially important for forest regeneration in man-made clearings (Fujita and Tuttle 1991; Nyhagen et al. 2005). In West Africa, Thomas (in Fujita and Tuttle 1991) found that seeds dispersed by bats accounted for 90–98 per cent of the first seeds of woody plants to arrive in clearings.

3.4.3 Consequences of the loss of a keystone species

The four dominant keystone species that shape and maintain the forests of Christmas Island are most likely to be:

- the red crab
- the Christmas Island white-eye *Zosterops natalis*
- the Christmas Island imperial pigeon *Ducula whartoni*
- the Christmas Island flying fox.

The significance of the red crab has been documented and quantified in some detail (Green et al. 1993). Conversely the significance of the other three species (the Christmas Island flying fox, Christmas Island imperial pigeon and Christmas Island white-eye) has hardly been investigated. Nevertheless, some comparisons can be made. By their sheer abundance and biomass, crabs are likely to be the most important species on Christmas Island. However, they operate only near ground level, influencing nutrient recycling, seedling recruitment and water penetration of the soil, but may pollinate and disperse seeds only over short distances at best. Christmas Island white-eyes are also very abundant, have brush-tipped tongues, which are important in pollination (but only of diurnal flowers), disperse the seeds of very small-fruited plants, and could have a large influence on the population levels of some insects. Christmas Island imperial pigeons are abundant and swallow fruits whole, so they disperse many seeds, but only up to a certain size, and they do not pollinate. Christmas Island flying foxes swallow smaller seeds (<5 mm) whole (duplicating the role of Christmas Island imperial pigeons), but also carry larger fruits in their mouths, and probably pollinate many plants. In the tropical forests of Africa and Asia, *Pteropus* species are typically the only animals capable of providing long-distance dispersal for rainforest plants with seeds greater than three centimetres long (McConkey and Drake 2006). Other animals, including insects and forest birds, might also pollinate and disperse some species. However, considering the overwhelming evidence of the significance of *Pteropus* species wherever they have been studied, and the limited alternative pollinators and dispersers on Christmas Island, clearly the services of the Christmas Island flying fox in the Christmas Island ecosystem must be unique and significant.

Banack (1998) showed that about 79 per cent of canopy-forming trees in Samoa are at least partially dependent on flying foxes for pollination or dispersal. Cox and Elmqvist (2000) warned that the loss of flying foxes from some Samoan Islands was affecting the reproductive biology of some plants, although they did not quantify the impacts. Richards (1990) proposed a ‘raiders and residents’ model of seed dispersion by flying foxes, whereby intense competition for fruit leads to dominant individuals defending food resources; less dominant individuals therefore raid territories and make off with stolen fruit, and consequently disperse seeds over long distances. The model predicts that, at low densities, flying foxes would cease to disperse larger seeds over long distances. McConkey and Drake (2006) demonstrated that at high abundance *Pteropus tonganus* effectively dispersed a large number of seeds long distances from food trees, but at low densities this dispersion process ceased. Thus, although *P. tonganus* survives at low densities, its ecosystem services do not persist.

Castings and dropped seeds from Christmas Island flying foxes were not found during this study. It is likely that the Christmas Island flying fox has already reached densities that are too low for it to continue functioning as a long-distance disperser of large seeds on Christmas Island. Plant species that could be affected include *Planchonella nitida, Terminalia catappa* (the favoured nesting trees of the endemic Christmas Island frigatebird *Fregata andrewsi*) and possibly *Barringtonia racemosa*. The extinction of the Christmas Island flying fox could cause major disruption to the pollination of numerous canopy trees. Changes in forest structure and floristics would have cascading effects that would bring about significant impacts on the substantial endemic biodiversity of Christmas Island.

3.5 Assessment of threats

Numerous species of flying foxes around the world are regarded as threatened (Mickelburgh et al. 1992; IUCN 2006), and the family itself is often considered particularly vulnerable. The literature lists only four identified threats to flying fox species around the world (IUCN 2006). These are:

- human predation and persecution (including culling to protect crops)
- habitat loss and fragmentation
- natural disasters (mostly cyclones)
- a single case of a predator (*Boiga irregualris*) invading camps on Guam (Cox and Elmqvist 2000).

A combination of factors has most likely caused the decline of the flying fox on Christmas Island (for example, a combination of habitat loss and over-harvesting followed by the eruption of yellow crazy ants with their direct and indirect impacts).

The present plight of the Christmas Island flying fox requires that its ‘proximal’ threat(s) be identified and mitigated promptly. Further discussion of available information appears in the following pages.

3.5.1 Predation

One or more predators could have caused the decline of the Christmas Island flying fox population. Introduced predators have been implicated in the decline of many vertebrate species, especially on oceanic islands, and Christmas Island has a suite of introduced predators.

Yellow crazy ant *Anoplolepis gracilipes*

Yellow crazy ants might directly disturb, displace or kill the Christmas Island flying fox. The scale and sooty mould that the ants induce have the potential to reduce fruit and pollen production and even kill food trees. In some areas, the expansion of the yellow crazy ant population might explain some of the decline in the population of the Christmas Island flying fox. However, there is a fairly poor geographical relationship between ant densities and the fate of camps. Ant densities have been very high at times in the vicinity of all camps, but especially high at Hosnies Spring, McMicken Point and

Greta Beach where camps persist. The temporal correlation between the increase in yellow crazy ants (late 1980s to early 1990s) and the decline of the Christmas Island flying fox (late 1980s to early 1990s) is very close. If yellow crazy ants have contributed to the decline of the Christmas Island flying fox, then a more substantial investment in control of yellow crazy ants could lead to a rise in the population of the bat.

Wolf snake *Ophites aulicus capucinus*

The BMP examined the distribution, abundance and diet of the wolf snake in 2005. A dietary analysis found no conclusive remains of mammals or birds; and behavioural observations suggest that the wolf snake is a ‘sit-and-wait’ predator that ambushes prey, and not an ‘active-search’ predator that would seek out and invade roost sites. Distribution data indicated that the wolf snake occurs in the west of the island, and enters primary forest at least sometimes. Abundance data were inconclusive.

Feral cat *Felis catus*

Feral cats have been implicated in the decline of wildlife in numerous habitats all over the world, particularly on islands (for example, Kirkpatrick and Rauzon 1986; Potter 1991; Low 1999; Lindenmayer and Burgman 2005). They are capable of climbing trees and would be able to smell roosts from considerable distances. In 1988, analysis of gut contents found that the flying fox occurred in 11.5 per cent of cat stomachs and made up 21 per cent of the diet by volume of feral cats on Christmas Island (Tidemann 1989; Tidemann et. al. 1994). However, they were not recorded in stomach or cat samples by Van der Lee (1997), while in 2002 ‘the presence of flying foxes was too few to make any firm conclusions’ (Corbett et al. 2003).

It is thought that Christmas Island flying foxes mostly fall prey to cats when feeding close to the ground in the introduced cherry *Muntingia calabura* (Tidemann 1989; Tidemann et al. n.d.). Van der Lee considered it was not possible to determine whether cats actually preyed upon flying foxes or merely scavenged injured animals and discarded waste after hunts. There are no current data on the distribution and abundance of feral cats across the island. However, Tidemann et al. (n.d.) and Algar and Brazell (2005) stated that they have an island-wide distribution. A review of information suggested that they probably occur island-wide, inhabiting undisturbed as well as disturbed areas for many years (Tidemann et al. n.d.). Densities are much higher than on the Australian mainland. Although cats could plausibly have a deleterious impact on the population of the Christmas Island flying fox, they probably do not account for the marked decline that occurred in the 1990s.

Nankeen kestrel *Falco cenchroides*

The nankeen kestrel was self-introduced to Christmas Island in about 1950 (Stokes 1988). Historically, nankeen kestrels have been observed harassing Christmas Island flying foxes. However, the temporal and geographical correlations between the decline of the Christmas Island flying fox and the establishment of nankeen kestrels on Christmas Island are not strong. In 2005, the birds were found to be distributed widely over the island and to be fairly common in suitable habitats. However, it is not clear when their population and distribution expanded. In 1974, van Tets (1975) estimated that there were no more than 100 pairs, and Stokes (1988) used the same figure without revision.

Therefore, it is possible that the nankeen kestrel remained at low abundance and predominantly in the north-east of the island until the late 1980s. If this were so, then the pattern would correlate better with the geographical and temporal patterns of decline in the flying fox. Nevertheless, nankeen kestrels (115–225 grams: Marchant and Higgins 1993) are much smaller than Christmas Island flying foxes (300–500 grams: Tidemann 1985). Nankeen kestrels feed predominantly on terrestrial invertebrates, and small reptiles, birds and rodents. The mammals recorded in their diets are limited to the Lesser Long-eared Bat *Nyctophilous geoffreyi*, a small insectivorous bat, rodents and a rabbit kitten (Marchant and Higgins 1993). They are not efficient at catching flying vertebrates, and eat a much smaller proportion of large vertebrates on Christmas Island than the Christmas Island goshawks do (BMP unpubl. data). It is highly implausible, due to the size disparity, that nankeen kestrels could have a significant impact on the abundance of the Christmas Island flying fox.

Christmas Island goshawk *Accipiter [fasciatus] natalis*

Tidemann (1985) noted that a flying fox was observed being eaten by a Christmas Island goshawk, *Accipiter [fasciatus] natalis*. Goshawks have been observed in all areas where the bats have been found. Goshawks have also been observed landing in trees where Christmas Island flying foxes were roosting, without harassing the bats or causing any apparent alarm (G. Dale pers. obs.).

In 2004, analysis of goshawk pellets found no evidence of Christmas Island flying fox remains (Hurley 2005). However, in September 2005, a juvenile female was observed attempting to catch a flying fox in flight at Margaret Knoll before dusk on two consecutive evenings. The goshawk did strike a couple of bats with its talons but did not manage to catch any. The goshawk and the Christmas Island flying fox have both been on the island for a long time, and there has been no marked increase in goshawk numbers, so any sudden decline in bat numbers is highly unlikely to be due to goshawk predation.

Humans

Accounts of island residents exerting pressure on local animal populations by hunting them for food were formerly numerous (for example, Andrews 1900; Strout 1939; Gibson-Hill 1949; Nelson 1972; Cogger et al. 1983; Stokes 1988; Neale 1989; Adams and Neale 1993). Newsome (1975) explicitly remarked on how island residents regularly killed flying foxes for food. In 1984, Tidemann (1985) attempted to estimate the level of human hunting pressure on the Christmas Island flying fox by distributing a questionnaire. The three questionnaires that were returned indicated that catches of 200 or so Christmas Island flying foxes might be common, but the frequency of hunting could not be estimated. It was thought that hunting could have been occurring since soon after settlement. Tidemann's (1985) research on population size and structure of the Christmas Island flying fox suggested that the impact of human predation was small.

Hunting of the Christmas Island flying fox is now prohibited. However, it is unclear when and why hunting ceased. In 1984, most island residents (wrongly) thought that hunting was banned (C. Tidemann in lit.). Legal protection was probably first given when the *Environmental Protection Act 1986* of Western Australia was applied to Christmas Island by the *Christmas Island Act 1958* (Cwlth). However, this protection was not immediately enforced to a level that would end hunting. Van der

Lee (1997, p. 23) stated that flying foxes were ‘legally hunted until recently and a number of *Pteropus* are almost certainly still taken illegally on a regular basis’. Possibly, the economic prosperity that came with award wages in the late 1980s obviated the need for wildlife harvesting, or changes in human demographics in the 1980s and 1990s led to most hunters leaving the island. Perhaps a sudden population decline of the Christmas Island flying fox made hunting no longer worthwhile. Whatever happened, the Christmas Island flying fox is no longer hunted in large numbers on Christmas Island. There have been very few reports of poaching in recent years (M. Orchard pers. comm.), although it may still occur clandestinely on rare occasions.

Whether hunting ceased before or after the apparent population crash of the Christmas Island flying fox is not known for certain. In fact, the exact date is not known for either event, although they occurred within a few years of each other. Even if hunting pressure did lead to the initial decline of the Christmas Island flying fox, it does not explain why the population has not rebounded. In the Samoan archipelago, the population of *P. samoensis* increased following a ban on hunting (Brooke 2001). Therefore, another threat or threats to the flying fox must currently be in operation.

3.5.4 Ecological health

The following hypotheses all concern some fundamental change in the ecosystem other than the establishment of predators.

Loss of habitat

A number of flying fox species have become threatened or extinct due to losses of roosting areas (Duncan et al. 1999; Flannery and Schouten 2001; Low 2003). About 25 per cent of Christmas Island was deforested between 1895 and 1987, but since then no clearing has occurred (Director of National Parks 2002). It is difficult to estimate the nett loss of habitat for the flying fox because many cleared areas have been colonised with introduced food sources, and the compensatory benefits of these are not known. There is no evidence of any historical clearing at campsites, and certainly there has been none since the time of Tidemann’s first study in 1984. The flying fox feeds on at least 35 different plant species, many of which are widespread (Table 15). The cyclonic winds of 1988 destroyed some habitat, but this should have recovered by 2006. The flying fox was recorded island-wide and in all habitats during this BMP. Even if a 25 per cent decrease in habitat occurred over almost 100 years, this is unlikely to lead to a sudden population crash of around 75 per cent.

Flying foxes in Australia tend to be limited by food supply rather than predation (P. Eby in lit.). It is possible that habitat loss has led to a change in the regularity of crucial food supplies. The evidence is not available to critically assess this hypothesis.

Cyclones and storm events

Cyclones have been implicated in the decline of other *Pteropus* species on tropical islands elsewhere (Pierson et al. 1996; McConkey and Drake 2006; IUCN 2006). Corbett et al. (2003) proposed that cyclonic winds swept most of the population of the Christmas Island flying fox out to sea on the night of 27 March 1988 and temporarily destroyed the food crop for the remainder. Thus, they proposed that the population crashed to current levels or lower overnight, and winds destroyed all of the camps. They compiled ‘anecdotal evidence provided by long-time residents’ of the island to support the hypothesis (Corbett et al. 2003, p. 79). Some of that evidence is circumstantial and some is misleading. For instance, there is an anonymous quote that the ‘flying foxes were sent to sea’. It seems unlikely that anyone actually witnessed such an event. A tropical low with gale-force winds did pass Christmas Island on that date, but the extent of damage it caused appears to be overestimated by Corbett et al. (2003). Other anecdotal evidence compiled as part of this BMP indicates that a noticeable decline in the Christmas Island flying fox occurred in the mid-1990s. Furthermore, there is no mention of a 1988 decline in Tidemann et al. (n.d.), a report that was still being worked on at least as late as 1993. However, that report did state that 100 Christmas Island flying foxes were counted at the McMicken Point camp in November 1988 by one of the authors (H. Yorkston), which was eight months after the storm event. It is nonetheless plausible that the 1988 tropical low did reduce the population. On its own, this hypothesis does not account for a subsequent decline in the 1990s, nor does it explain why the population has not recovered 18 years later.

Poisoning by Fipronil®

The only chemical known to have been distributed in remote and forested areas on the island to any significant degree is Fipronil®, the poison used in the yellow crazy ant baiting program since 2000 (Green et al. 2004). Spreading bait granules over the forest canopy during extensive helicopter baiting in September 2002 (Green et al. 2004) could potentially have exposed the flying fox to high doses of Fipronil® which they ingested while feeding. The potential for non-target impacts of Fipronil® are not entirely known. A few studies on Christmas Island have detected few impacts on vertebrate or invertebrate species (Stork et al. 2003; Marr et al. 2003), although experiments to date have had limited scope. Fipronil® was not used on Christmas Island until 2000, and not delivered by helicopter until September 2002 (Green et al. 2004), which is well after the decline of the Christmas Island flying fox first occurred.

Light pollution

The presence of lights might potentially disturb an insular forest bat to the degree that it abandons traditional foraging areas. However, Christmas Island flying foxes have been recorded foraging around lights in the settled area of the island. Furthermore, whilst light pollution might explain a shift in distribution, it cannot account for a sudden decline in abundance. Lights on Christmas Island are mainly restricted to settled areas, where they have been present for many decades, and so light pollution could not account for a decline in abundance across the entire island.

3.5.5 Disease and parasites

There is no evidence of disease or parasite loading in the Christmas Island flying fox population. In 1984 Tidemann found no internal parasites in the Christmas Island flying fox and two species of ectoparasite were collected. However, these were found only on single animals out of well over 100 individuals sampled (Tidemann 1985). Tidemann et al. (n.d.) also noted that the flying fox was remarkably parasite-free. There have been no assessments of disease and no recent assessments of parasite loading, so it remains plausible that one or both of these have increased since 1984.

3.6 Legal protection

3.6.1 Protected status

The Christmas Island flying fox is not listed as a threatened, migratory or marine species under the EPBC Act. The species probably first gained legal protection through Western Australian laws.

The Action Plan for Australian Bats (Duncan et al. 1999) listed the Christmas Island flying fox as ‘data deficient’. The assessment was based on Tidemann’s studies from the mid-1980s. The flying fox is currently listed as ‘Lower Risk (Least concern)’ by the IUCN (2006) but was last assessed in 1996. It is currently under consideration for listing as threatened under the EPBC Act.

3.6.2 Protected areas

Christmas Island is managed by Parks Australia for environmental protection and biodiversity conservation in accordance with the EPBC Act. About 63 per cent of Christmas Island forms the Christmas Island National Park. Two of the three remaining flying fox camps are in the national park: Hosnies Spring and McMicken Point. The Hosnies Spring camp is also in the Hosnies Spring Ramsar Site. The Greta Beach camp is in unoccupied Crown land. Flying foxes frequently forage and sometimes roost singly or in small groups in the Dales Ramsar Site. Both Ramsar sites are entirely inside the national park. Probably about 25 per cent of potential foraging habitat and potential roosting habitat is outside the national park. It is not known what proportion of the population roosts outside the park or leaves the park to forage, and how important habitat outside the national park is to the species’ survival.

3.6.3 Current management actions

Environmental management aimed at conserving biodiversity on Christmas Island consists mainly of rehabilitation of minefields, control of invasive ants, and control of invasive plants. There are no specific management actions in place or currently proposed for the conservation of the Christmas Island flying fox.

4. Land birds

During 2005, the BMP established baseline data sets of relative abundance for most of the island’s forest bird species. This is necessary because of the high rates of endemism and conservation significance in the bird fauna of Christmas Island, and considering the high level of stress generally facing the biodiversity of Christmas Island.

Six of Christmas Island’s endemic forest bird species – one self-introduced species and a forest-nesting seabird – were recorded in this survey. Appendix B contains more detail about the survey methods.

4.1 Summary of results

The Christmas Island white-eye was the most abundant species with an overall reporting rate of over 99 per cent (Table 16). The Christmas Island goshawk was the least abundant of the eight target species with a reporting rate of just over 1 per cent.

The Golden Bosun *Phaethon lepturus fulvus* was less common in semi-deciduous forests than in cleared areas. Nankeen kestrels were recorded more frequently in cleared habitats than forested ones. The Christmas Island Imperial Pigeon was more common in evergreen forest than in either semi-deciduous forests or cleared areas. The Emerald Dove *Chalcophaps indica natalis* was recorded less frequently in cleared habitat than in evergreen forest. The Island Thrush *Turdus poliocephalus erythropleurus* was more frequent in evergreen forest than in disturbed habitats. The Christmas Island white-eye was uniformly distributed across all levels of each factor considered in the analyses.

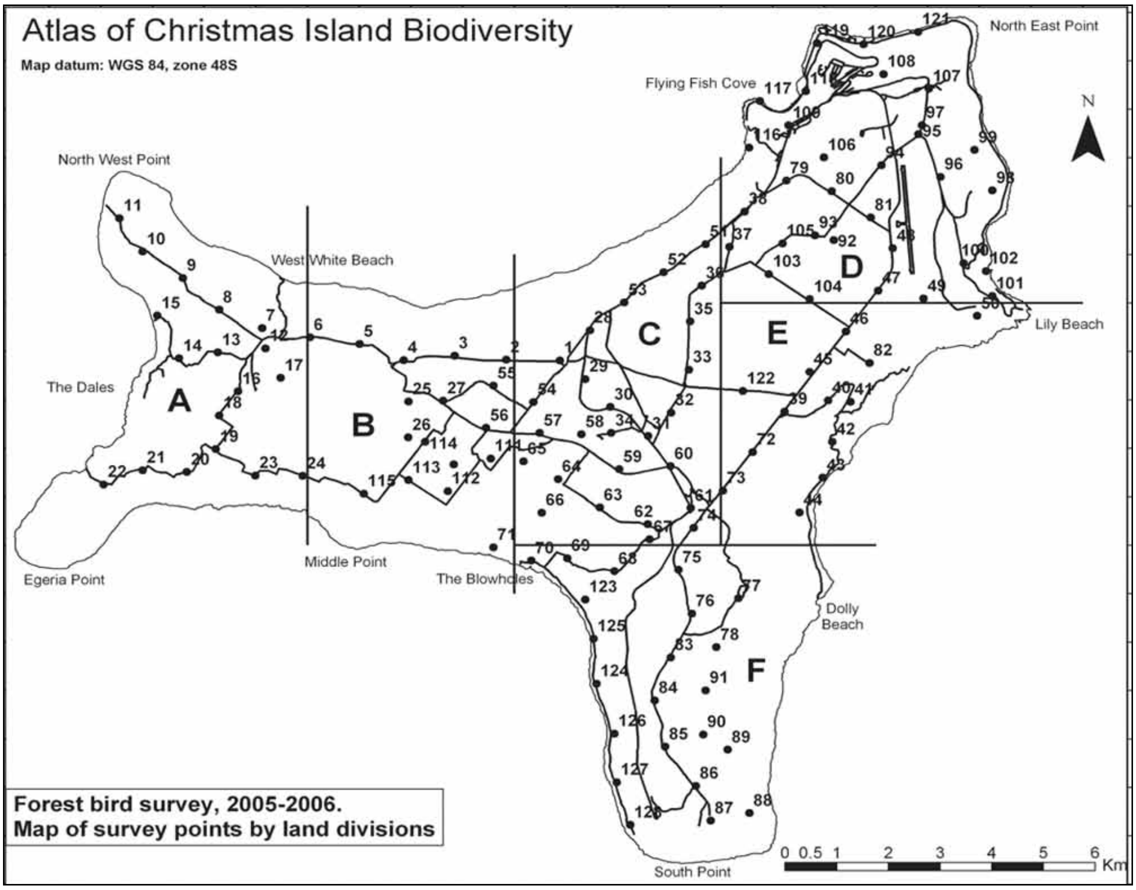
This survey suggests that most of the native forest bird species of Christmas Island are secure. Exceptions might be the Christmas Island goshawk and the Christmas Island hawk-owl *Ninox natalis*. This survey provides a baseline data set that can be used in the future to determine trends in most of the species of forest birds on Christmas Island. The ability to analyse the data by habitat and sector of the island (amongst other potential variables) could well provide insight into the cause of any trends detected in the future.

Table 16: Counts and reporting rates of forest birds by habitat

	Bosun	Goshawk	Kestrel	Pigeon	Dove	Thrush	Swiftlet	White-eye
Habitat A	41	3	9	132	65	110	61	130
(n = 132)	31.06%	2.27%	6.82%	100.00%	49.24%	83.33%	46.21%	98.48%
Habitat B	23	1	13	98	45	79	57	103
(n = 104)	22.21%	0.96%	12.50%	94.23%	43.27%	75.96%	54.81%	99.04%
Habitat C	106	1	13	98	45	79	57	103
(n = 182)	43.62%	0.41%	35.80%	87.65%	30.45%	59.67%	65.84%	99.18%
Habitat D	16	1	8	29	14	22	24	32
(n = 32)	50.00%	3.13%	25.00%	90.63%	43.75%	68.75%	75.00%	100.00%
All habitats	**186**	**6**	**117**	**472**	**198**	**356**	**302**	**506**
(n = 527)	36.26%	1.17%	22.81%	92.01%	38.60%	69.40%	58.87%	98.64%

Figure 20 shows the sample sites in relation to sectors

Figure 20: Sectors of the island used for geographical analysis of forest bird abundance



4.2 Christmas Island goshawk

The Christmas Island goshawk *Accipiter [fasciatus] natalis* is endemic to Christmas Island and is listed as endangered under the EPBC Act. It is also the island's top predator in the terrestrial environment. No surveys of the goshawk have been undertaken previously, and no baseline data on the population existed. The BMP funded a month-long banding expedition by the Australasian Raptor Association in August 2004 (see also Hurley 2005). See Appendix B for a description of the methods.

The survey results suggest a total population size of around 250 birds. This calculation does not take into account factors such as mortality, biases and so forth, so it must be considered as approximate.

The results of the dietary analysis are presented in Table 17. Until this survey, goshawks were generally considered to feed mainly on birds. The most surprising findings were that:

- all pellets contained a large proportion of the giant grasshopper *Valanga irregularis*
- insects made up an estimated 82 per cent of prey items in the sample
- scales of a small skink (probably grass skink *Lygosoma bowringii*) were found in one pellet.

Table 17: Prey items in nine Christmas Island goshawk pellets

Prey item	No. of pellets	% of pellets	No. of items	% of total
Birds	4	44%	4	16%
Reptiles	1	11%	1	2%
Grasshoppers (*Valanga irregularis*)	9	100%	5	79%
Beetles (Coleoptera)	2	22%	1	2%
Mantids (Mantidae)	1	11%	41	11%
Total Insects	*9*	*100%*	*47*	*82%*

1 = number of pellets containing the prey item; 2 = percentage of pellets containing the prey item; 3 = total (minimum) number of individuals of prey item in total sample; 4 = approximate percentage of prey item in total sample.

4.3 Christmas Island hawk-owl

Hill (1996) established a baseline on the abundance of the Christmas Island hawk-owl *Ninox natalis* and set out a methodology for repeating the first survey to obtain a population trend. It requires 18-minute call-playback surveys at 22 sites across the island, repeated four times each, with surveys spaced a few weeks apart at a minimum. The BMP attempted to repeat Hill's original survey. One round of surveys was completed in early 2005, but the second survey was abandoned due to heavy rains. Other operational requirements and unfavourable weather conditions meant that the survey could not be completed in 2005.

4.4 Island thrush

In December 2005, two intern students (L. Olsen and J. Murakami) undertook a short study of the nesting biology of the island thrush *Turdus poliocephalus erythropleurus*. This endemic subspecies is listed as endangered under the EPBC Act, and the other two subspecies in Australia (at Lord Howe and Norfolk Islands) are both extinct. The only existing information on the subspecies was through general natural history observations. The students were given a list of 52 specific questions, the answers to which would describe the nesting biology in detail. They were instructed to:

> find as many nests as possible and observe or measure as many different aspects of breeding biology as possible. Because nests are fairly easy to find and breeding is loosely synchronised, it should be possible to study several breeding efforts at the same time (for example, some building nest, some with eggs, some with fledged chicks) and thereby cover many different stages of breeding in a relatively short period of time.

Two study sites were established: one at the Pink House Research Station and one at Territory Day Park and surrounding areas.

Six nests were found, with three nests at each study site. Nests were made almost exclusively of palm fibre; other materials, such as leaves, were incorporated only rarely. All three nests at the Pink House were located in birds-nest ferns. One nest at the Territory Day Park site (along the nature trail) was located in a tree hollow. Nests appeared to be higher and nesting birds more wary inside the forest compared with outside. Nests appeared to be located towards one end of a territory. Territorial disputes were common at the boundary between territories.

Four of the six nests had clutches of two eggs and one nest had two chicks. The second egg was laid about three days after the first, incubation commenced with the laying of the second egg, and the eggs hatched on the same day. In one nest, only one egg hatched, and the other was left in the nest. Adults removed egg shells from the nest by picking them up in the bill and flying off with them. They then ate the egg membrane but not the shell. The incubation period was 10 to 13 days. Changeover of incubation was observed only once. It seems that the incubating bird leaves the nest and then the other parent arrives to continue the incubation. Adults gave a long, thin drawn-out whistle monotonously while incubating, but also gave the whistle away from the nest.

The brooding period was about 10 days. It could not be ascertained whether adults brought water to chicks at the nest, but they did appear to 'feed' while not carrying anything visible in the bill. Chicks called very little from the nest, most of the time.

Once chicks left the nest, they did not return to it. Juveniles were dependent for at least a week and probably a little, but not much, longer. Adults had a small, moving territory around dependent juveniles when foraging with them, and they attacked other adults that entered that territory.

At one nest the adult pair had a dependent juvenile and was also incubating another clutch of two eggs. The juvenile would occasionally approach the nest and beg the incubating bird to feed it. This made incubation slightly irregular. On one occasion, the incubating adult did not respond immediately to the begging juvenile, so the juvenile pushed the adult out of the nest. Once the new eggs hatched, the adults ignored the juvenile and the juvenile stopped begging them, but the juvenile continued to forage (quietly) in the territory. This indicates successive nesting attempts in a season; the nesting season comprises:

- courtship (unknown time frame)
- nest building (unknown time frame)
- laying (2–3 days)
- incubating (10–13 days), brooding (10 days)
- post-fledging dependency (7–12 days)
- successive clutches could be laid as frequently as 25–30 days apart.

Island thrush foraged almost exclusively on the ground and used perches only as lookout posts and for roosting. They have also been seen hunting geckos on a shadecloth awning and hunting insects on window sills and building walls, so presumably they sometimes forage in trees. The most common item in the diet was millipedes, but beetles were also taken frequently.

They avoided pandanus *Pandanus tectorius*, an observation also made by Stokes (1988). No predation was observed. An observation of five juveniles in the same tree may represent a crèche (unlikely) or flocking after the dependency period.

5. Seabirds

Seabirds were surveyed as part of the BMP because some nest only on Christmas Island, with significant nesting sites not far from the IRPC.

5.1 Abbott’s booby

Abbott’s booby *Papasula abbotti* is a seabird that nests only on Christmas Island (Stokes 1988) and is listed as endangered under the EPBC Act. The population declined significantly during the 1960s to the 1980s due to habitat loss as a result of mining. The first declaration of national park on Christmas Island was specifically to protect Abbott’s booby (Director of National Parks 2002). The species nests in the canopy of the island’s forests, and monitoring programs conducted in the 1980s indicated that breeding success was significantly reduced when nests were located downwind of clearings (Reville et al. 1990).

There have been five surveys of the breeding population of Abbott’s booby since settlement. These were in 1967 (Nelson 1971), 1979–80 (Powell and Tranter 1981), 1982 (Nelson and Powell 1986), 1991 (Yorkston and Green 1992), and 2002 (Olsen 2004).

The 1991 survey was undertaken from the ground and was the most comprehensive and accurate survey. It was estimated that there were 2,679 breeding pairs in 1991 (Yorkston and Green 1992). It is no longer physically possible to repeat the 1991 survey according to those who were involved in it because it is now much harder to navigate in the forest, much more difficult to move across the ground, and harder to see into the canopy than it was in 1991. The reasons are that the old mine grid lines from the 1970s that were used as a grid system (about 300-metre grid) have since overgrown and the understorey of the forest has thickened over large areas with the changes due to yellow crazy ants.

The 2002 survey was conducted by helicopter over two days during the aerial baiting campaign for yellow crazy ants. Ideally, it would have been groundtruthed immediately (by sub-sampling), but following the aerial baiting all resources were employed surveying the effectiveness of the baiting, assessing the impacts on crab populations, and hand-baiting localised populations. The helicopter survey was briefly reviewed by Olsen (2004, p. 25). She considered it to be ‘a time-efficient, if not cost-efficient, means of future monitoring’. She also claimed that the 2002 ‘survey can act as a baseline’. She considered that the 1,500 nests counted in 2002 compared favourably with the estimate of 2,679 breeding pairs in 1991 (Yorkston and Green 1992). This statement needs to be treated cautiously as two different survey methods were used, and it was not possible to calibrate the aerial survey data with ground observation. Olsen said that ‘To the north of the IRPC there is a considerable density of nesting boobies that should be considered in any activity that might cause their disturbance’. She also noted that Abbott’s boobies appeared to have recolonised some areas of forest downwind of minefields where revegetation buffers were approaching 20 years old.

The BMP undertook surveys of the number of nests of Abbott’s booby, adjacent to the IRPC and in control sites nearby, in the 2004, 2005 and 2006 nesting seasons. The aim was to monitor for early signs of potential impact from the construction of the IRPC, such as reduced nesting density in the area or movement of nests way from the forest edge adjacent to the IRPC. Methods are described in Appendix D.

Nest counts: results and discussion

Nest counts are presented by survey plot and year in Table 18 and Figure 21. The number of nests recorded increased slightly from year to year in most of the measured parameters (see Yorkston and Green 1992 for a discussion of interpreting the signs of nesting), but it is possible that survey effectiveness increased slightly during the duration of the study. In 2004, all nest trees required tagging, whereas, in subsequent years, only new trees required tagging, reducing the effort required for the survey.

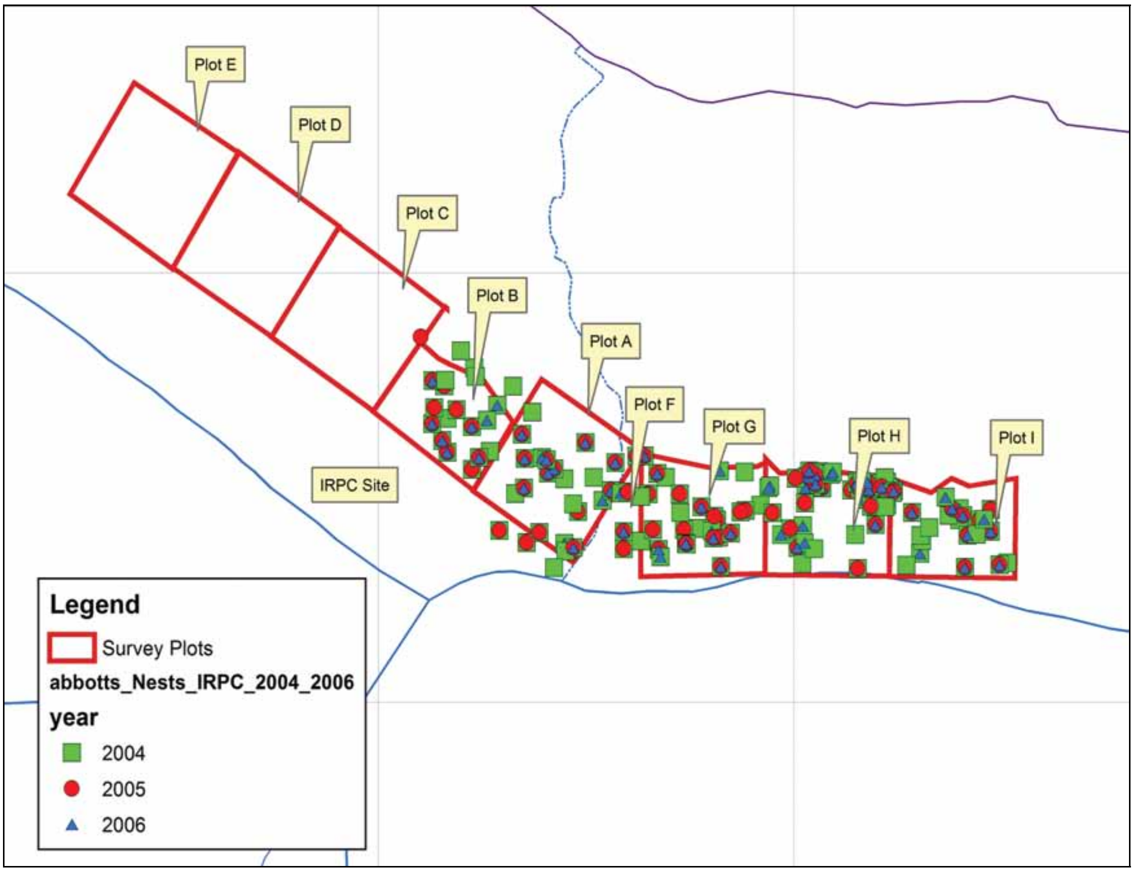
The survey was designed to detect extreme changes. There was no measurable change in the density and distribution of Abbott’s booby nests in the vicinity of the IRPC between 2004 and 2006, and no strong indication of any movement of nests away from the IRPC, with the possible exception of three to four nests located between the southern edge of Plot A and the forest edge near the front gate of the IRPC.

Table 18: Summary of Abbott’s booby nests recorded by survey plot and year

Year	Plot	Total nests	Birds	Male	Female	Juvenile	Nest	Chick	Guano
2004	A	24	13	0	2	1	7	0	24
2004	B	20	7	0	0	0	9	0	21
2004	F	7	6	0	0	1	3	0	7
2004	G	26	16	0	0	0	16	0	30
2004	H	49	24	0	0	2	26	1	51
2004	I	25	16	1	1	1	13	0	30
2005	A	27	16	0	2	0	13	0	23
2005	B	21	12	0	1	0	10	0	20
2005	F	6	3	0	1	0	4	0	6
2005	G	30	29	0	2	0	18	0	32
2005	H	54	37	0	3	0	27	0	48
2005	I	23	14	2	1	1	10	0	20
2006	A	38	29	8	2	1	24	3	33
2006	B	17	12	6	1	0	14	2	17
2006	F	8	4	0	0	0	3	0	8
2006	G	30	23	3	6	1	25	1	31
2006	H	49	43	13	12	8	39	0	42
2006	I	25	21	5	3	3	18	0	17
2004	ALL	151	82	1	3	5	74	1	163
2005	ALL	161	111	2	10	1	82	0	149
2006	ALL	167	132	35	24	13	123	6	148

Note: The figures in the ‘Total nests’ and ‘Birds’ columns refer to the total number of nests and birds recorded in each plot area. Where a bird could be positively identified as a male, female or juvenile that has been shown in the ‘Male’, ‘Female’ or ‘Juvenile’ column. The number of birds recorded in nests in the plot area is shown in the ‘Nest’ column. The number of sites in the plot area where guano was recorded is shown in the ‘Guano’ column.

Figure 21: Map of Abbott's booby surveys near the IRPC site



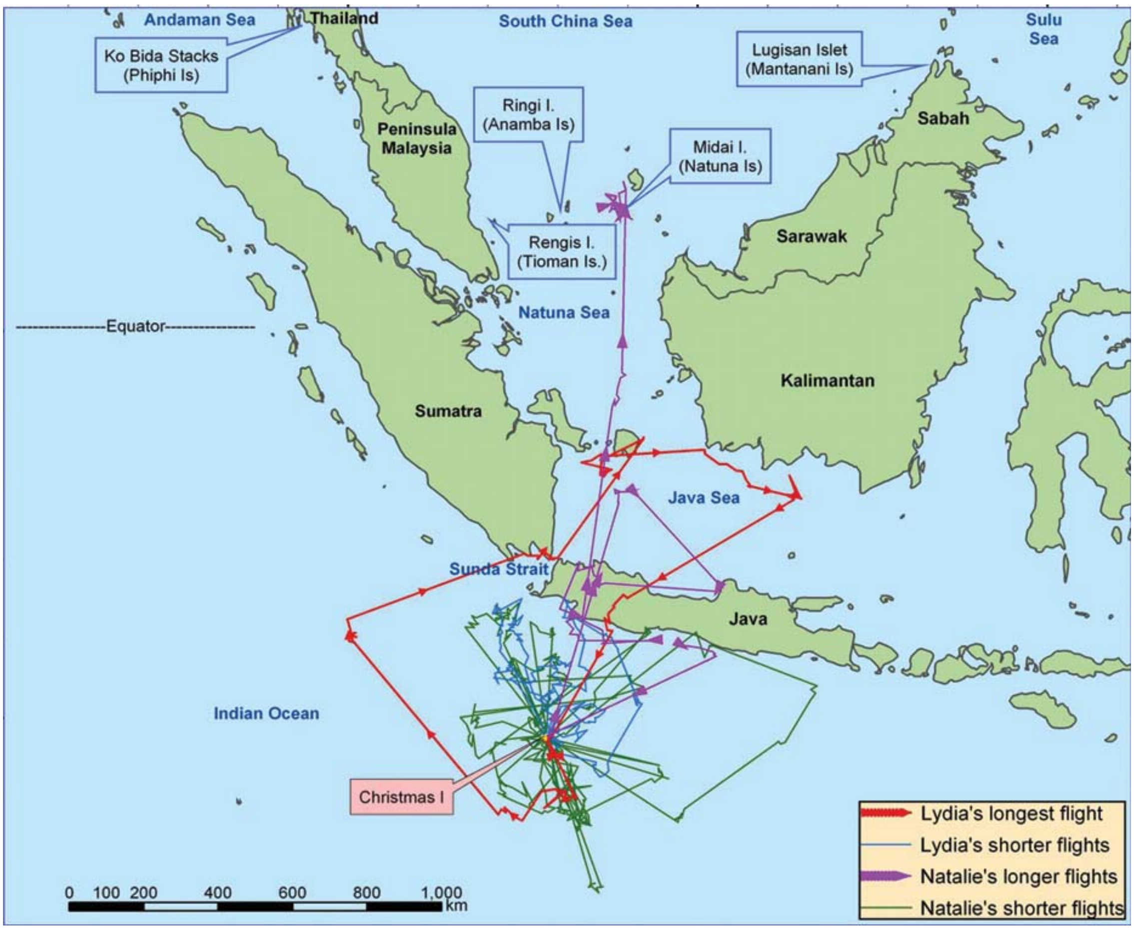
5.2 Christmas Island frigatebird

A detailed survey was undertaken to map the nests of the Christmas Island frigatebird *Fregata andrewsi*, and assess its population size, population trend and nesting success in 2004 and 2005. The methods followed those established by James (2003) and aimed to build on the existing database compiled in 2003. Surveys were undertaken at the height of the breeding season in April 2004 and April 2005, and towards the end of the breeding season in 2004. An additional nesting colony was located on the shore terrace of the northern coast, near the eastern edge of the national park boundary, well west of the dust fall-out zone from the phosphate dryers. About 100 pairs nested there in 2004 and 2005.

James (2003) could not identify any significant threats on Christmas Island that might explain the gradual decline in the population size of the Christmas Island frigatebird. It was speculated that threats might be operating in the species' marine habitat, which includes extensive areas in Southeast Asia. To assess this scenario, satellite tracking devices were attached to two breeding female frigatebirds in 2005. The devices were awarded through a grant program run by the American Bird Conservancy and NorthStar Science and Technologies, Inc. The experiment was conducted by the BMP in collaboration with Dr Janos Hennicke from the University of Hamburg.

Initial results indicated the extensive range of the frigatebird. Dr Hennicke's research program is continuing. Figure 27 shows the tracks recorded for the two birds between September 2005 and May 2006.

Figure 22: Flights of two Christmas Island frigatebirds recorded with satellite transmitters between September 2005 and May 2006



5.3 Red-tailed tropicbird

Three successive internship projects supervised by the BMP studied the breeding biology of the Red-tailed tropicbird *Phaethon rubricauda* in 2005 and 2006. Nests at two colonies in the settlement were tagged, and frequent visits to the nests allowed compilation of extensive nest diaries. Adult birds were marked on their plumage with unique colour combinations of sheep-crayons for individual identification. Data were collected on the length of incubation and brooding periods, pair fidelity, nest site fidelity, growth rate of chicks, duration of incubation and foraging stints, and nesting success. More than 200 nests were observed and over 5,600 nest observations were made between September 2005 and July 2006. In 2006 the nesting success rate was extremely low (less than 5 per cent). Chicks were disappearing from their nests and it appeared that predators were involved. Sand-trapping failed to unambiguously identify predators. A report by the last of the three intern students summarises the results up to July 2006 (Ishii 2006).

The project was subsequently continued by a German MSc student under the supervision of Dr Janos Hennicke of the University of Hamburg. A remote camera supplied by the BMP was used to identify feral cats preying on two chicks and a black rat preying on another.

5.4 Seabird banding

Bird banding projects were established to augment other research being conducted on seabirds on Christmas Island. An institutional project licence was issued to the Government Conservator of Christmas Island by the Australian Bird and Bat Banding Scheme (ABBBS) for this purpose. Five subprojects were authorised under the licence for Christmas Island goshawks, Abbott’s boobies, seabirds, island thrushes, and Java sparrows. The banding studies are ongoing and require annual reporting to the ABBBS on the number of birds banded and inventories of ABBBS bands held in stock.

6. Reptiles

Reptile studies by the BMP began with a wide-ranging survey in 2004, followed by a number of more specific studies of selected species and groups.

In the first half of 2004, an extensive survey was undertaken to map the distribution of native and introduced reptiles across the island. Summaries of relative abundance statistics for reptiles from the quantitative surveys are given in Table 19.

Lister’s gecko *Lepidodactylus listeri* and the Christmas Island blind snake *Typhlops exocoeti* are both listed as vulnerable under the EPBC Act and are two of the five endemic reptiles occurring on Christmas Island. Neither species was located in the survey. Both species have not been recorded since the mid-1980s (Cogger 2005), and concerns about their status were raised by Cogger and Sadlier (2000). This study has confirmed that these two species are of concern. Full assessment of their status is made difficult by their cryptic behaviour.

At this point the processes that threaten them are unknown. Since the BMP reptile survey was completed, a national recovery plan has been prepared for these two species (Cogger 2005). The BMP survey made an important contribution to the recovery plan. A brief review of records of the Christmas Island blind snake was subsequently prepared by the BMP (see Section 6.6, below).

Table 19: Relative abundance of reptiles as recorded from the 2004 reptile survey

Species	Number of sites where recorded	Total count	Recording rate per hour
Forest skink	0	0	0
Blue-tailed skink	7	34	0.129
Coastal skink	0	0	0
Grass skink	52	149	0.567
Lister’s gecko	0	0	0
Christmas Island giant gecko	79	159	0.605
Barking gecko	27	77	0.293
Asian house gecko	1	3	0.011
Christmas Island blind snake	0	0	0
Flowerpot blind snake	16	17	0.065
Asian wolf snake	2	2	0.008

6.1 Summary of results

Surveys in 1979 (Cogger et al. 1981, 1983) and 1998 (Cogger and Sadlier 2000) provided maps of species distributions, some abundance data based on the number of specimens collected, and general observations on the species’ abundance and distribution. The BMP survey was the first quantitative baseline survey of terrestrial reptiles for Christmas Island. The results show declines in the five endemic reptiles, and increases in most introduced reptiles.

Two of the five endemic reptiles are missing (not recorded since the 1980s), namely the Christmas Island blind snake and Lister’s gecko. The native but not endemic coastal skink is facing local extinction. The endemic blue-tailed skink and forest skink have both declined and are facing a severe risk of extinction: these have both significantly contracted in distribution in recent years and are now restricted to a handful of small, isolated pockets on the coastal terraces, and extending only narrowly behind the sea cliff edge. Likely reasons for their declines are intense predation and competition from infestations of the invasive yellow crazy ant. This was confirmed during the BMP (Gorge 2005).

One endemic gecko remains common and widespread: this is the giant gecko *Cyrtodactylus sadleiri*.

In contrast, the same survey revealed that four of the five introduced species are common and widely distributed, and the fifth is also common but with a limited distribution.

6.2 Forest skink

The endemic forest skink *Emoia nativitatis* has declined severely. It was not recorded in the quantitative surveys, although skinks were sighted during the qualitative surveys that led us to conduct further searches. Qualitative surveys specific for forest skinks (10–15-minute surveys with one observer creeping quietly into position and waiting silently and motionless for skinks to emerge in sunny patches) were undertaken at several locations and forest skinks were eventually recorded at five localities. All locations were on the shore terrace in semi-deciduous forest or semi-deciduous vine thicket, with limestone pinnacles. The largest populations were on the western and eastern coastal terraces at South Point. The population below Tom’s Ridge, near the IRPC, was very small, only a few individuals were recorded at North West Point on the coastal terrace, and only one individual was recorded at Middle Point. Reports of forest skinks by Parks Australia staff and associates since the survey have come from Egeria Point, the West White Beach Track, Taman Sweatland, south of the golf course, the resort and Martin Point. Follow-up surveys at these locations have generally found grass skinks to be common, and no further populations of forest skinks have been found. Even so, further surveys on Egeria Point would be worthwhile.

It appears that the forest skink is now confined to scattered, localised pockets in remote areas of the coastal terraces and first inland cliff. Further colonies may exist, but they are likely to be small and few in number. In 1979 the forest skink was widespread in the forests and considered to be the most abundant reptile on Christmas Island (Cogger et al. 1983). In 1998 it was recorded less frequently, but this was attributed to overcast and wet weather, which reduces their activity, so it was not considered that they were threatened (Cogger and Sadlier 2000). It is possible that they had already begun to decline by 1998. Reasons for the decline are unknown. The timing of the outbreak and spread of yellow crazy ants suggests a possible link, but the geographical pattern of the ant outbreak does not correlate well. Other factors, for example, the spread of poison ant baits, cannot be ruled out. Feral cats are likely to be a major predator, as native reptiles have been found in their stomach contents on several occasions (Corbett et al. 2003). They probably pose significant threats to the small and isolated colonies that remain.

The ecology of the forest skink, along with the blue-tailed skink, was studied in more detail along the Boulder Track in 2005 and 2006 by intern students attached to the BMP (Gorge 2005; Rueff and Bordelet 2006).

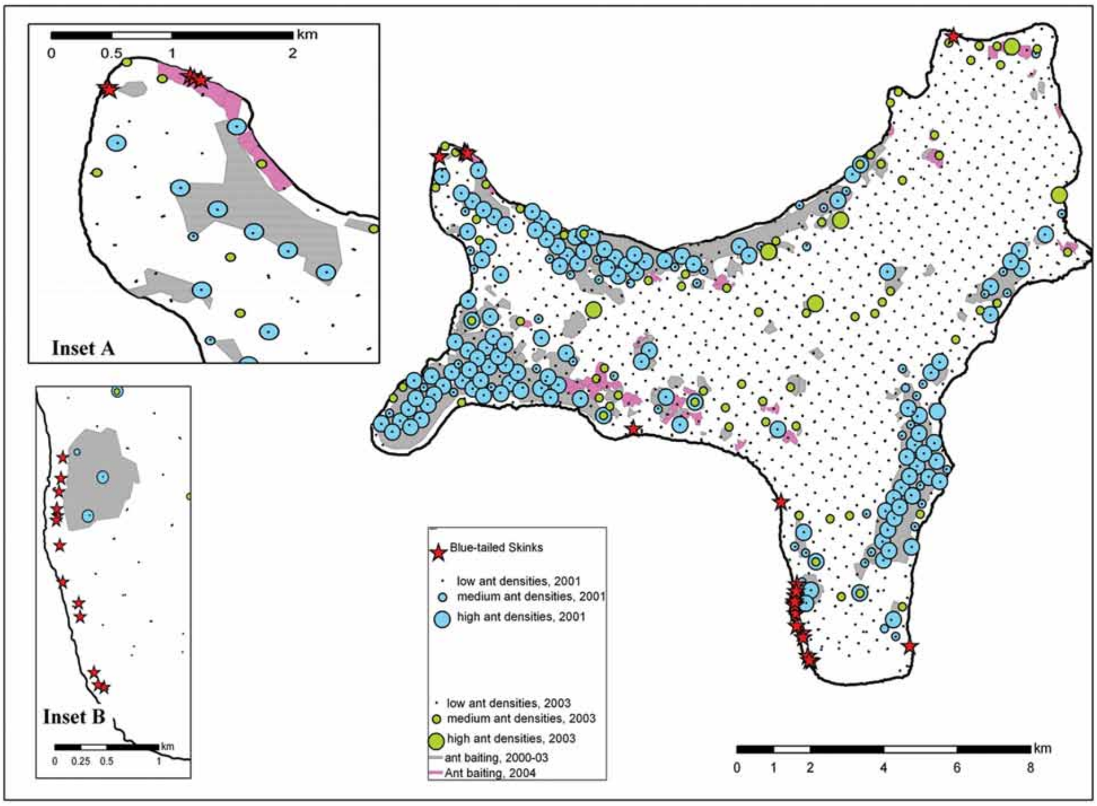
6.3 Blue-tailed skink

The endemic blue-tailed skink *Cryptoblepharus egeriae* has also declined severely and the pattern appears to be very similar to that of the forest skink. The blue-tailed skink now appears to be confined to scattered, localised pockets in remote areas of the coastal terraces. No populations were found close to the IRPC during this BMP. This skink was described as abundant and conspicuous in 1979, although its distribution was recorded as patchy (Cogger et al. 1983). It was most abundant in settled areas of the shore terrace (Settlement, Flying Fish Cove), but was also found on fallen trees in plateau rainforest and coastal forest sites.

Its decline and disappearance from settled areas were first noted by Rumpff (1992), who recorded it in the diet of wolf snakes. Rumpff, and subsequently Cogger and Sadlier (2000), assessed the species as highly threatened. Rumpff postulated that predation by the introduced wolf snake and the self-established nankeen kestrel could have caused the decline, although other factors could be involved. Cogger et al. (1983) reported that the species was common in the pinnacles of abandoned minefields, but they gave no specific localities. This habitat was searched on about 20 occasions at different locations between 2004 and 2006 (searches not included in effort statistics reported above) but no skinks were seen. Blue-tailed skinks have not been reported in mine pinnacle fields since 1979.

A plausible but untested hypothesis is that this brightly-coloured species is vulnerable to predation from nankeen kestrels on the exposed pinnacles. This is an arboreal (tree-climbing) skink, and Cogger et al. (1983) speculated that it might dwell in the canopy of plateau rainforest, although no one has been able to survey that habitat. Richard Hill (pers. comm. 2004) found blue-tailed skinks basking on a tree platform in the canopy that he erected near the West White Beach track in 1996.

Figure 23: Map of the distribution of blue-tailed skinks, yellow crazy ants and Fipronil®



6.3.1 Survey results

The potential for Fipronil® baiting affecting skink populations was noted and further investigated. Using Parks Australia's GIS, data were compared on the distributions of:

- yellow crazy ants (collected during island-wide surveys in 2001 and 2003 by Parks Australia's Invasive Species Team)
- Fipronil® baiting (collected from 2000 to 2004 by Parks Australia's Invasive Species Team)
- remnant blue-tailed skink populations (collected by the BMP in 2004).

Blue-tailed skinks were recorded during seven quantitative surveys (34 individuals, 0.129 records per survey hour). Qualitative surveys specific for blue-tailed skinks (similar to those used for forest skinks) recorded the species at several other localities.

The largest colony of the blue-tailed skink was located along the boulder track. The next largest colony was found near the south-eastern end of South Point (Medwin Point), but it was much smaller. Blue-tailed skinks were seen here regularly between July and August in 2004 and 2005, but none was seen during the same months in 2006, when feral cats were thought to be more abundant (J. Hennicke, pers. comm.). It seems likely that this colony is now extinct. Reports of forest skinks by Parks Australia staff and others since the survey have come from Egeria Point and single records from a garden in Drumsite and a garden in Silver City. Follow-up surveys at Egeria Point and searches at Martin Point have not located this species, but further surveys would be worthwhile. Further monitoring of blue-tailed skinks on the coastal terrace of North West Point in late 2004 and early

2005 revealed small and scattered pockets of skinks spread out over a few kilometres of coast, and extending only narrowly behind the sea cliff edge.

The 2004 survey did not involve canopy searches, so it remains unknown whether the species (still) occurs in plateau canopy.

6.3.2 The effects of Fipronil® on blue-tailed skinks

In April 2004 the BMP team discovered a small remnant population of blue-tailed skinks on the top of the sea cliff at North West Point. Moderate densities of yellow crazy ants were also present. The Invasive Species Team at Parks Australia indicated that they were intending to bait the area with Fipronil® in late 2004. This provided an opportunity to experimentally assess the impact of both yellow crazy ants and Fipronil® on blue-tailed skinks. The survey methods are in Appendix E.

Blue-tailed skink populations continued undiminished for more than a year in the baited areas whereas yellow crazy ants were eliminated (Table 20). This study has demonstrated with sufficient certainty that it is safe to use Fipronil® for hand-baiting in the vicinity of blue-tailed skinks in the field. The decline of ants after baiting confirms that the Fipronil® reached its target and the experiment reflects typical baiting regimes implemented by the Invasive Species Team. There was no indirect effect that could be attributed to deprivation of arthropod prey induced by Fipronil®.

Table 20: Summary statistics for blue-tailed skink counts at North West Point

Statistic	All plots	Coastal	Interior	Before	After
N	96	48	48	48	48
Sum	423	315	108	180	243
Mean	4.4	6.6	2.3	3.8	5.1
Std deviation	4.34	4.86	2.24	3.73	4.83
Maximum	27	27	10	17	27
Minimum	0	0	0	0	0
Zero counts	12	1	11	10	2

6.3.3 Monitoring endemic skinks at South Point

Two intern projects were supervised by the BMP to investigate the ecology of the blue-tailed and forest skinks on the western shore terrace of South Point, along the Boulder Track (Gorge 2005; Rueff and Bordelet 2006). The location was chosen because the largest populations of both species occur there together and the access is reasonably good. The survey methods are described in Appendix E.

Gorge (2005) found that the presence of yellow crazy ants affected the distribution of blue-tailed skinks, and that the distribution of the forest skink was heterogeneous. Blue-tailed skinks were more common in the coastal zone and forest skinks were more abundant in the inland zone of the study site. There was otherwise no significant effect of habitat on skink distribution. The decline of the species was found to be almost certainly the result of introduced species such as the wolf snake and feral cat.

Rueff and Bordelet (2006) confirmed that there were no significant correlations between the presence of skinks and habitat parameters. To determine the distribution and abundance of skinks on the boulder terrace, a survey of the entire terrace was conducted. A north–south gradient was found in the abundance of skinks along the terrace with higher numbers in the south. Blue-tailed skinks had a more coastal distribution than forest skinks. Although three species occurred together, each seemed to prefer slightly different microhabitats; they occurred together with limited interaction because they occupied different strata.

6.4 Coastal skink

The native (but not endemic) coastal skink *Emoia atrocostata* is confined to the rocky littoral zone of the island. The techniques employed in this survey were not suitable for assessing either the status or distribution of the coastal skink. No quantitative surveys were undertaken in the species habitat. A few species-specific surveys were undertaken in areas where they are known to have been seen in recent years. Only one individual was located, on the wide coastal terrace at Middle Point. Possible sightings were also made on the western terrace of South Point and at the Blowholes boardwalk. The coastal skink was present at the mouth of Dale No. 3 in 2002 and at Winifred Beach in 2003 (D. James, pers. obs.), and it was reported twice from the latter location in 2004 and 2005. Opportunistic searches were made at Middle Point, the Blowholes, Flying Fish Cove, Lily Beach, Ethel Beach, Low Point, Medwin Point and the western shore cliff of South Point (Smithson Bight) during seabird surveys and other visits between 2002 and 2006 (Parks Australia, unpubl. data). The coastal skink was not recorded in any of these localities.

Cogger et al. (1983) reported the coastal skink to be very confiding and also highly active, which would make it quite easy to detect. They recorded the species at Flying Fish Cove, Lily Beach and Smithson Bight. It appears that the coastal skink has declined severely and is likely to be on the verge of extinction on Christmas Island. A plausible but untested hypothesis may be vulnerability to predation by nankeen kestrels on the exposed pinnacles of the coastal cliffs.

6.5 Giant gecko

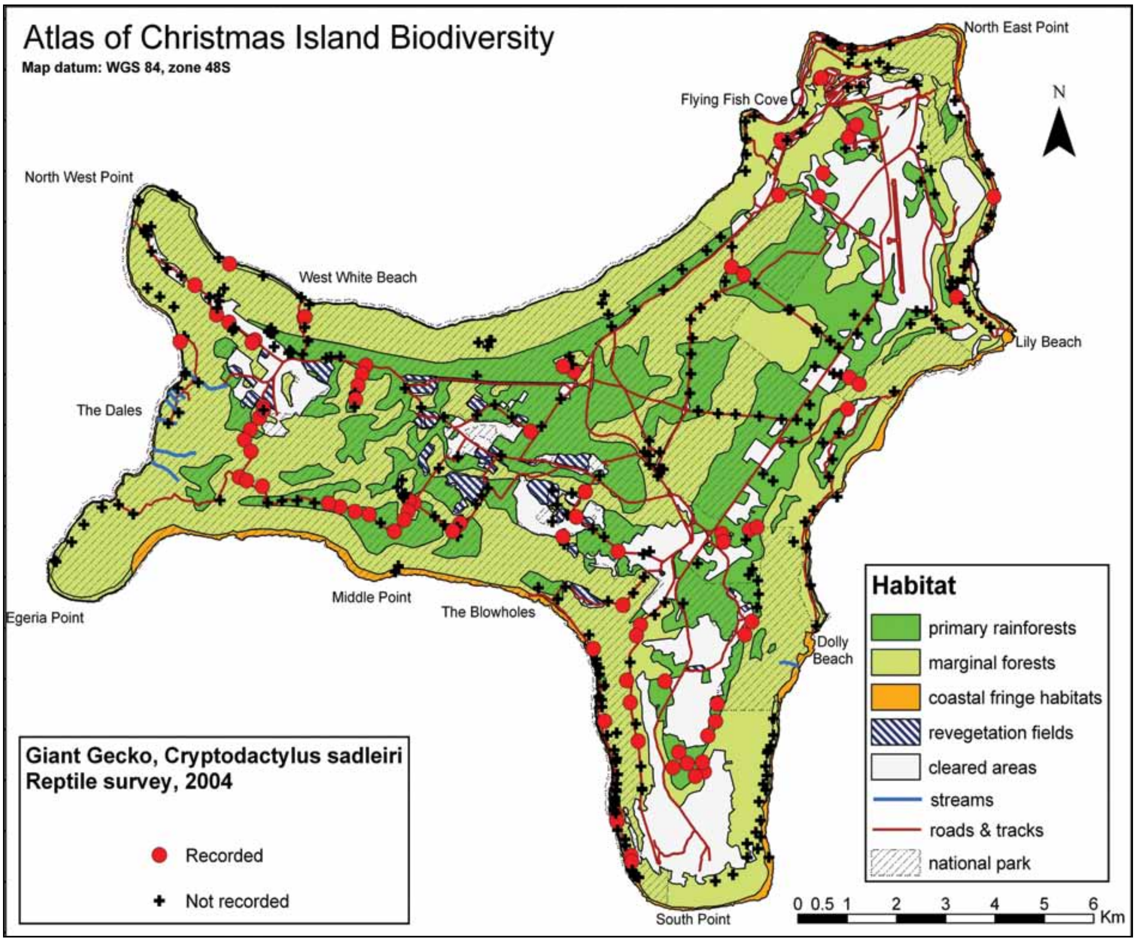
The endemic giant gecko *Cyrtodactylus sadleiri* remains common over a large part of the island. It was the most frequently recorded and widely distributed species documented in the survey (Figure 24). It is the only native reptile known to still occur in forests immediately adjacent to the IRPC.

A total of 159 individuals was recorded at 79 of 420 sites (0.605 sightings per search hour) during the quantitative diurnal surveys. In a limited number of nocturnal surveys it was recorded at higher densities.

On many occasions, the species was recorded in areas with low or medium densities of yellow crazy ants, and was found inside logs that also contained yellow crazy ant nests. This suggests that it has some tolerance to yellow crazy ant infestations. The species was recorded more frequently on the plateaus than on the shore terraces. Some localised declines might have occurred. It was not recorded in the block of evergreen tall closed rainforest on the central plateau between Hanitch Hill and the

Pink House in 2004. It was recorded on the Hanitch Hill (or Plateau) Track during spotlighting surveys in February 2006, but at very low densities. This area of the plateau was not surveyed extensively in either the 1979 or 1998 studies, although the giant gecko was previously common in the vicinity of the Pink House (H. Cogger, pers. comm. 2004). This is one of the areas least affected by yellow crazy ants (Parks Australia unpubl. data).

Figure 24: Distribution of the giant gecko as recorded in the 2004 reptile survey



6.6 Christmas Island blind snake

The Christmas Island blind snake *Typhlops exocoeti* has not been recorded since the mid-1980s. The species has been documented only a handful of times and it is very poorly known. This review attempts to compile the scant information of relevance to its abundance and distribution to better inform conservation efforts.

6.6.1 Review of previous reports

HMS Flying Fish

In January 1887, the *HMS Flying Fish* spent a few days anchored at Flying Fish Cove, and made a brief landing at West White Beach, but landing parties were unable to explore beyond their landing points (Maclear 1887; Gibson-Hill 1949). During this time they collected two Christmas Island blind snakes. Although these specimens have no precise locality data, they can have originated only from Flying Fish Cove. The snakes were 350 millimetres long by 6 millimetres diameter and 230 millimetres long by 3.5 millimetres diameter. They were described collectively as ‘pale brownish, each scale with a brown spot; these spots largest and darkest on the dorsal surface, where they form longitudinal lines’ (Boulenger 1887, p. 517).

HMS Egeria

HMS Egeria spent about 10 days anchored at Flying Fish Cove in October 1887. Parties reached the plateau and were able to explore more widely than previous expeditions (Lister 1888). A single specimen of Christmas Island blind snake was collected, but no precise locality is available.

Andrews

Andrews (1900) spent 18 months on Christmas Island in 1897 and 1898. He collected ‘several’ specimens that were found ‘in damp places, under rocks and fallen trees’. The specimens were again described as pale brownish with spots, but the ‘total length’ was given as 480 millimetres.

Gibson-Hill

Gibson-Hill (1947) spent 26 months on Christmas Island between September 1938 and November 1940. He described this snake as fairly common, but his description of ‘young snakes’ being almost black indicates that *Ramphotyphlops braminus* was present at that time and he was confusing the species. His description of them being pale brown with dark spots is almost identical to Boulenger’s earlier descriptions. He described a ‘typical’ specimen as 372 millimetres. He gave no specific localities, and did not say how many specimens he collected.

Subsequently, Tweedie of the Raffles Museum identified two typhlopids collected by Gibson-Hill as *R. braminus* (Cogger and Sadlier 1981). D. James examined Gibson-Hill’s typhlopid specimens at the Raffles Museum, Singapore, in 2004. Only two were present, both labelled as *R. braminus* in Tweedie’s handwriting. Both specimens were clearly Christmas Island blind snakes. They were both about 200 millimetres long, brownish-orange in colour and lacked the distinctive oil glands present between the scales of the head in *R. braminus*. It is unclear how many specimens Gibson-Hill collected, though there were at least two.

Cogger and Sadlier

Hal Cogger and Ross Sadlier undertook dedicated reptile surveys in 1979 and 1998. They did not find any examples of the Christmas Island blind snake. During the 1979 survey, they were presented with a single specimen taken in 1975 at Field 22, near Stewart Hill (Cogger and Sadlier 1981; Cogger et al. 1983).

Records in the 1980s

One Christmas Island blind snake was captured in a pitfall trap in 1985 by Nick Dunlop (pers. comm.) at ‘Field 22 South in Plateau Rainforest on the red soil’. A photo of this individual appeared in the 1986 Christmas Island calendar. Only one Christmas Island blind snake was recorded in 200 to 300 trap days, although there were many captures of forest skinks and giant geckos. Another turned up during reclearing of Field 17, probably in about 1986 (Hugh Yorkston, pers. comm.). David Powell apparently had some records during the 1960s and 1970s. The two individuals that Hugh Yorkston saw in the 1980s (Fields 22S and 17) were all pink in colour and pale, about one foot (about 30 centimetres) long and as thick as a little finger. Those held for a few days did not change colour.

Recollections of Kim Chey

According to Kim Chey (pers. comm., 2004), Christmas Island blind snakes were occasionally encountered during the 1970s and 1980s in the forest, wrapped around pinnacles basking, particularly on sunny afternoons after heavy overnight or morning rain. No specific localities or dates were provided. They were described as steely grey, about the thickness of a little finger and resembling a piece of rope wrapped around the pinnacle.

Shrew surveys

In 1997–98 Paul Meek (2000) sampled 820 pitfall trap nights searching for shrews. No Christmas Island blind snakes were caught. The only reptiles caught were five giant geckos.

Assessment of effectiveness of different trapping survey techniques

Corbett et al. (2003) summarised the effectiveness of survey methods for reptiles following their 26-day fauna survey in August 2002. These data are collated in Table 21.

Table 21: Summary of trapping effectiveness in August 2002

Method	Effort	Effectiveness
Diurnal search surveys	8.35 am	61 reptiles at 7.3 reptiles/hr 5.9/hr for giant gecko; 0.2/hr for *Gehyra* and *Ramphotyphlops braminus*; 0.1/hr for *Hemidactylus,* and 0.1/hr for *Lygosoma*
Spotlighting	< 10 hrs	65 reptiles at 6.6 reptiles/hr 5.7/hr for giant gecko; 0.1/hr each for *Hemidactylus*, *Gehyra* and *Lycodon*
Cat scats	92 scats	2 forest skinks, 1 *Lycodon aulicus*
Pit-traps	366 days	7 *Lygosoma bowringii*, 1 giant gecko
Elliot traps	1,252 nights	1 giant gecko

6.6.2 BMP reptile survey during 2004

A total of 320 quantitative searches were made for reptiles across Christmas Island in the first half of 2004. The total effort was 262 hours 40 minutes of searching (see Appendix E for a description of the methods). During the survey, the microhabitat of the Christmas Island blind snake, as gleaned from the scant published accounts, was sampled in unprecedented levels, but not one was found. Altogether, 225 reptiles (0.86 individuals per hour) were recorded.

7. Land crabs

Crab mortality on the roads of Christmas Island was monitored for most of the duration of the BMP. The sex and size-class structures in the populations of both red crabs *Gecarcoidea natalis* and robber crabs *Birgus latro* were also examined.

Land crabs drive important ecological processes on Christmas Island and significantly influence the nature of the forests. The land crabs of Christmas Island are both intrusive on and susceptible to human activities. Large numbers are killed by traffic every year despite intensive management efforts by Parks Australia. This report examines aspects of the susceptibility of red crabs and robber crabs to activities associated with the construction of the IRPC.

Data on the population structure and road mortality of these two species are presented.

7.1 Summary of results

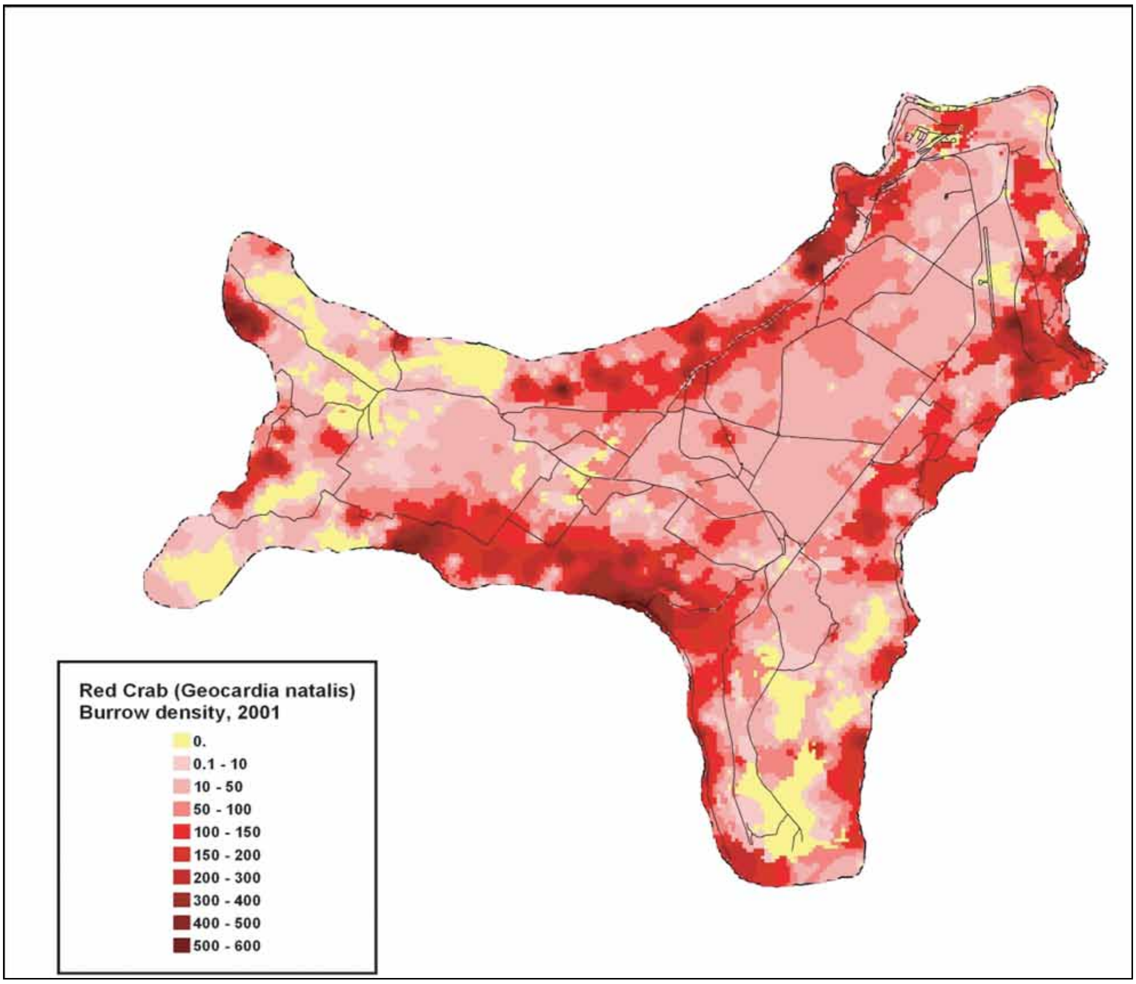
Following extensive research into the timing and routes of red crab migration, Morris and Adamczewska (n.d. [circa 1996], p. 47) predicted without qualification that 'Increased use of the Irvine Hill and Murray Roads during the migration season will increase crab mortality'. That prediction has proved correct. Modelling from the 2001 island-wide survey data clearly indicated that red crabs occur in high density along Murray and North West Point Roads – routes that were well-used during construction of the IRPC. Because crabs are active whenever relative humidity is high at ground level, it is clear that these roads will contain high numbers of crabs on a regular basis. Modelling also predicts extremely high levels of crabs crossing these roads during migration periods.

The IRPC construction process has caused high levels of crab mortality. The location of the IRPC on the north-western end of Christmas Island, far from the accommodation and port facilities in the north-east, has inevitably led to a substantial increase in traffic on the island's roads. In particular, the traffic on the route between Drumsite and the IRPC via Murray and North West Point Roads has increased as a result of the construction of the IRPC. These two roads run through areas of high density of red crabs (Figure 25) and robber crabs. Where vehicle traffic comes into contact with crabs, it is typical that substantial numbers of crabs are killed (Hicks 1983; Morris and Adamczewska n.d.).

Red crabs play a fundamental role in shaping and maintaining the globally unique forest ecosystems of Christmas Island (O'Dowd and Lake 1989, 1990, 1991; Green et al. 1993). The health of their population is inextricably tied to the health of the island's ecosystems and biodiversity. The population of robber crabs on Christmas Island appears to be the only healthy one remaining in the world (Hicks et al. 1990).

The populations of concern are of very high conservation significance at a global level, and were already under considerable stress. Planning processes for the location of the IRPC made insufficient use of available ecological information. The *post hoc* development of an environmental management plan failed to mitigate the impact. There is, however, considerable evidence from these results for the effectiveness of crab management procedures (fencing and crossings) when they are properly implemented.

Figure 25: Map of modelled red crab densities on Christmas Island



7.2 Population size and structure: red crabs

Red crabs are the most numerous, widespread and ecologically important of the terrestrial crabs on Christmas Island (Morris and Adamczewska n.d.; Green et al. 1993; Gray 1985). The species is endemic to Christmas Island and has intrinsic conservation values, as well as being a significant resource for science, tourism and journalism (Morris and Adamczewska n.d.; Parks Australia 2002). Red crabs have frequently featured in magazine articles and wildlife documentaries around the world.

7.2.1 Previous estimates of population characteristics

The abundance and biomass of red crabs on Christmas Island are extraordinary. Estimates of the entire population size have varied from 32 million to 100–120 million (Morris and Adamczewska n.d.; Hicks 1985; also Hicks et al. 1990). Green et al. (1993) estimated an average of 1.3 crabs per square metre and 1,450 kilograms per hectare in some parts of the island (Green et al. 1993); this exceeds estimates of biomass per unit area in the Amazon basin for all animals combined (Green et al.

1993). Most recently, the results of a Parks Australia 2007 survey estimated a population size of 50 million red crabs.

The influence that red crabs have on the unique structure, characteristics and floristics (plant composition) of Christmas Island's forests is profound (O'Dowd and Lake 1989, 1990, 1991; Lake and O'Dowd 1991; Green et al. 1993; Parks Australia 2002). The crabs are omnivorous and opportunistic, feeding on green and dead leaves, fruits, seeds, seedlings, carrion and some animals. Through the differential predation of fruit, seeds and seedlings they influence the species composition of plants in Christmas Island forest and provide biotic resistance to invasive weeds. They dominate nutrient recycling in the ecosystem by burying and consuming 39–86 per cent of leaf litter on the forest floor, which influences growth rates of plants and the composition of invertebrate assemblages. They also prey on and control the invasive giant African land snail *Achatina fulica*. It is likely that their burrowing influences the rates of rainwater permeation and dehydration of forest soils, although this has not been studied. Thus, the habitat and survival of a range of other species, many of them endemic, are linked to the ecology of the red crab (Parks Australia 2002). The removal or decline of red crabs would produce a dramatic effect on the forest ecology of Christmas Island (Green et al. 1993), and it would very likely result in numerous adverse cascade effects.

Although they are terrestrial, red crabs are vulnerable to desiccation. They are almost exclusively diurnal, but their activity level is dependent on high levels of relative humidity (Hicks 1985; Green et al. 1993). High levels of activity on the ground generally occur when the relative humidity is 85 per cent or greater at ground level (Green et al. 1993). Distinct seasonal and diel patterns in relative humidity on Christmas Island mean that surface activity of red crabs is high in the wet season and low in the dry season, high in the morning, low in the middle of the day, and moderate to low in the late afternoon (Hicks 1985; Green et al. 1993).

Red crabs are highly susceptible to impacts from human activities (Morris and Adamczewska n.d.). The seasonal and diel patterns in their activity levels, and especially their mass migrations, bring large numbers of crabs into conflict with human activities. Despite the general predictability of conflicts between red crabs and human activities, planning processes on Christmas Island have so far taken little account of the ecology of red crabs.

At the present time, red crabs appear to be threatened by yellow crazy ants, road mortality, and possibly failures in recruitment (Parks Australia 2002). It is thought that predation by yellow crazy ants has reduced the total red crab population by 25–30 per cent during the 1990s and early 2000s (Parks Australia 2002).

7.2.2 Life cycle and the annual migration

The life cycle of red crabs is tied to the sea for spawning and the development of larvae (Hicks 1985; Morris and Adamczewska n.d.). This leads to mass annual migrations in which most of the adult population leave their territories in the forest and proceed to the shore where they mate, spawn, and return inland (Hicks 1985; Morris and Adamczewska n.d.). The migration is tied to the lunar cycle and the dates of spawning at the coast can be predicted within a few days, but the migration itself is triggered by rainfall and its duration is influenced by both rainfall and relative humidity (Hicks 1985; Morris and Adamczewska n.d.).

There is a pattern of episodic recruitment of young crabs. In some years, many millions of young crabs ‘return’ from the sea; in some years, moderate numbers ‘return’; but in most years (at least in recent times) there is no significant return (Hicks et al. 1990). The most recent substantial, island-wide return of baby crabs was in 1999 (M. Orchard pers. comm.), five years before the sample was collected. The effect on the age-structure of the population has never been investigated in detail.

Hicks (1985) showed that up to three migration sequences occurred each year in a breeding season lasting up to three months. Each migration sequence was characterised by:

- migration from the forest to the shore by adults
- dipping in salt water
- retreat to burrows on the shore terrace for copulation
- dipping by males followed by their return migration
- incubation of eggs by females in burrows
- movement of females to the shore for spawning
- return migration of females to the forest
- a marine larval stage
- return of baby crabs from sea to shore and their migration inland.

The timing of each sequence is linked to the lunar cycle, with spawning occurring at night on the turn of the high tide between the last quarter of the moon and the new moon (Hicks 1985).

7.2.3 BMP survey data

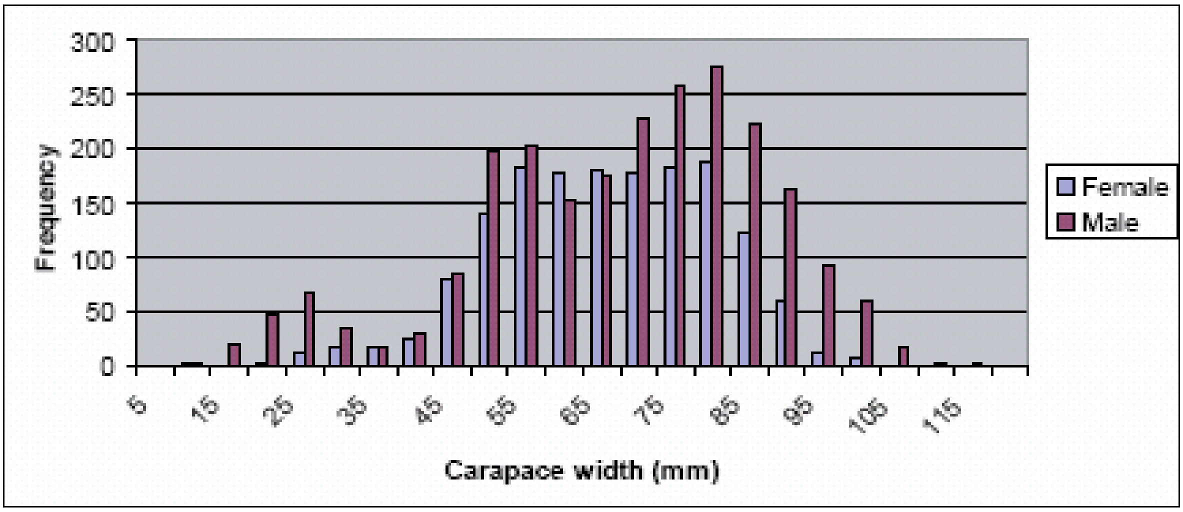
In 2004, the BMP sampled almost 4,000 red crabs to create a profile of the sex ratio and size structure of the red crab population, as this was not previously well recorded. Size was used as a surrogate for age because no way of reliably ageing crabs was available. The study was experimental and was not aimed to specifically test hypotheses, but rather to look at what insights could be provided by the data.

Table 22 presents summary statistics of size (carapace width) for males and females; the size distribution is in Figure 26. The low number of small crabs likely reflects sampling bias (smaller crabs are more difficult to find and capture) and failed recruitment over recently preceding seasons.

Table 22: Summary statistics of carapace width in 3,930 red crabs

Statistic	Male	Female	Both sexes
Sample size (n)	2,346	1,584	3,930
Mean (mm)	65.799467	63.8605	65.0121
Std dev (mm)	19.420655	14.3761	17.58797
Minimum (mm)	7.05	7	7
Maximum (mm)	112	100	112

Figure 26: Frequency histogram of carapace width in red crabs



The sample included 2,346 males and 1,584 females. The sex ratio was significantly skewed in favour of males by just under 3:2 (χ^2= 147.7, df = 1, p<0.00001; n=3,931). The sex ratio was not uniform across the island. At sites on the inland plateau and middle terraces, the sex ratio was 1:0, which was not significantly different from 1:1 (χ^2 = 3.8, df = 1, p=0.05004; n=1,350). Conversely, at sites on the coastal terraces, the sex ratio was significantly skewed in favour of males by 2:1 (χ^2 = 275.7, df = 1, p<0.00001; n=2,368).

Males in the sample were slightly but significantly broader across the carapace than females by an average of 1.94 millimetres or 2.95 per cent (two-tailed t-test: t = 3.39, df = 3,928, P < 0.0007).

Moderate recruitment occurred at some points around the island in early 2005 and early 2006 (Parks Australia, unpublished data). This could have produced a ‘spike’ in the size-class structure of the population corresponding to recruitments in 2005 and 2006 and a ‘trough’ corresponding to failed recruitments in the early 2000s. With future sampling, it might be possible to track the progression of those spikes and troughs, derive size-specific growth rates and ultimately calibrate the size-classes with age-classes. However, noise in the data and plastic growth rates of individuals will make this difficult.

The skewed sex ratio found in this study is difficult to explain. This study found that males outnumbered females by two to one on the coastal terraces, but the two sexes occurred in equal proportions at the sites higher than 90 metres above sea level. Morris and Adamczewska (n.d.) reported a different pattern in the mid-1990s: the lower terraces were occupied by female, sub-adult and juvenile crabs with relatively few males, and the proportion of males increased with altitude or distance from the shore. They located areas in the central plateau that were largely occupied by big males at low density and few if any females. In this survey we did not sample those plateau areas, but our samples on the coastal terraces were as large as those of Morris and Adamczewska (n.d.). Possible reasons for the difference could be the time of year in which sampling took place, or a change in the distribution of red crabs since the mid-1990s (for instance related to mortalities induced by yellow crazy ants). Further study is needed to confirm the precise reasons.

7.2.4 Baby crabs

A total of 179 baby crabs was measured as they returned to shore. They were an average of 3.7 millimetres (sd=0.19, min= 3.0, max=4.2) across the carapace. A total of 76 crabs that had been kept in a terrarium since they had first reached the shore were measured when they were 18 months old. The baby crabs raised in a terrarium grew to be about 16 millimetres on average after 18 months, but varied between 10 and 28 millimetres. If the growth rates approximate those attained under natural conditions, then it might be expected that crabs at the 50 millimetres mark are two to four years old (see Hicks et al. 1990). These observations lend some support to the observation that the last mass recruitment was in 1999.

A sex ratio is not available because red crabs smaller than 25 millimetres cannot be safely sexed by visual inspection of the abdominal shield. According to Hicks et al. (1994), abdomens of young female crabs do not broaden until after the first three years.

7.3 Population size and structure: robber crabs

The robber crab is probably the largest terrestrial arthropod in the world (IUCN 1981). Like red crabs, robber crabs have a marine larval phase and are therefore linked to the sea to complete their life cycle. Their migrations are much less conspicuous than those of the red crab, apparently because they are less abundant, only the females migrate, and their good climbing abilities mean that they can access remote parts of the coastline (George 1978; Gray 1985). Unlike the red crab, the robber crab is not endemic to Christmas Island, but has a wide distribution through the Pacific and Indian Oceans (IUCN 1981; Gray 1985; Davie 2002). The species was listed as globally ‘vulnerable’ in the 1981 Red Data Book (IUCN 1981) because in most places that are inhabited by humans the crabs have been hunted to extinction or nearly so. In 1996 they were downgraded to ‘data deficient’ (Eldredge 1996) – not because they had recovered but due to a lack of information.

8.3.1 Previous studies

The literature on robber crabs from Christmas Island is very sparse. In 1887 they were considered abundant and the most conspicuous invertebrate in the vicinity of Flying Fish Cove (Lister 1888). In 1897–98 they were found in abundance all over the island, and were largely diurnal (Andrews 1900). Harms (1933, cited by Rumpff 1979 in Parks Australia file archives) considered that Christmas Island was the only location where robber crabs were known to be diurnal, and reported that they were spread over the entire island, generally at an average of one per square metre.

By 1978 their distribution seemed uneven, and they apparently avoided certain areas, following heavy exploitation as food, bait and specimens for the tourist curio trade (George 1978). They also appeared to be more nocturnal than previously, and it was suggested that this might have related to the extinction of the two species of endemic rats (George 1978). About the same time, H. Rumpff (1979 file note in Parks Australia file archives) considered that the species had declined since the reports by Harms (1933) and Gibson-Hill (1947). The crabs were still fairly abundant on the central plateau, at Tom’s Ridge, and in the south-west, but apparently reduced in density on some of the lower terraces.

The crab was no longer strictly diurnal, but circadian, apparently due to increased disturbance (Rumpff 1979).

Hicks et al. (1990) reported a healthy population in 1979 with between 67 robber crabs per hectare on the central plateau and 166 red crabs per hectare at Egeria Point. They also reported that these estimates agreed with a subsequent survey by Schiller (1988, cited in Hicks et al. 1990). Since that time, yellow crazy ants have caused dramatic declines of robber crab abundance in the Egeria Point area (P. Green, pers. comm.).

Christmas Island is a significant global stronghold for robber crabs (Hicks et al. 1990). Consequently, the Christmas Island population has national, if not international, significance. They are important to Christmas Island's tourism industry.

At the present time, robber crabs appear to be threatened by yellow crazy ants, road mortality and persistent illegal poaching for food. Anecdotal evidence suggests that the population is declining gradually, but there are no reliable population estimates or trend data.

There are potential risks from the yellow crazy ant control program as a result of non-target impacts, although hand removal and using diversionary food sources before baiting reduce this risk.

7.3.2 Robber crab population structure

There is virtually no information on the current abundance, density or biomass of robber crabs on Christmas Island. In 2004, the BMP measured 538 robber crabs to create a profile of the sex ratio and size structure of the robber crab population (see Appendix F for methods). Size was used as a surrogate for age, because no way of reliably ageing crabs was available. The study was experimental and was not aimed to specifically test hypotheses, but rather to look at what insights could be provided by the data.

A total of 378 males and 160 females were measured, giving a significantly skewed sex ratio of about 2.36:1 ($\chi^2 = 88.3$, df = 1, $P<0.00001$). Summary statistics of measurements are given in Table 23. Males were much larger and more abundant than females.

Table 23: Summary statistics of size measurements from 538 robber crabs

Sex	Statistic	Carapace width (mm)	Carapace length (mm)	Weight (gm)
Male	n	378	348	360
	Mean	121.1	54.1	1,402.0
	sd	27.1	12.0	705.6
	min	25	12	55
	max	175	82.2	3,285
Female	n	160	159	157
	Mean	89.6	40.1	546.0
	sd	16.0	6.7	233.0
	min	41	20	65
	max	124	54.9	1,420

Sex	Statistic	Carapace width (mm)	Carapace length (mm)	Weight (gm)
Both sexes	n	538	507	517
	Mean	111.7	49.7	1,142.1
	sd	28.3	12.4	719.8
	min	25	12	55
	max	175	82.2	3,285

As with the red crab population, individual robber crabs in smaller size classes were relatively rare. This confirms probable sample bias: young robber crabs occupy discarded shells and so can be hard to find.

The reason for the low proportion of females is not known, but it could be a result of sample bias (young females can be hard to sex) and the possibility of the territorial nature of females and the wandering nature of males. This would have broad implications for conservation management because males would be killed more frequently by traffic on roads, and sedentary females would be highly susceptible to non-target poisoning from the yellow crazy ant baiting program. Not only does ant baiting occur predominantly inside the forest, but territorial animals would be less likely to be drawn away from baited areas by non-toxic lures. The hypothesis of territorial females and wandering males should be tested because of the potential for serious impacts on the female population of robber crabs that could be caused by existing management regimes, if it were true.

7.4 Road mortality of crabs in 2004 to 2006

Red crabs, robber crabs and other species cross the roads of Christmas Island, not just during migration periods, but at all times of the year when humidity levels permit crab activity. Crab activity on the roads can be considered either unpredictable or predictable. It is unpredictable if you want to know some days or weeks in advance what levels of crab activity on the roads can be expected at a specific time. It is predictable if it is accepted that crabs will be active on roads whenever humidity levels are above 70–80 per cent (Hicks et al. 1990; Green et al. 1993): this is whenever it is raining, in the early mornings and late afternoons, and at night. Further predictability is provided by the model of red crab distribution provided in Figure 25, which predicts high levels of red crab activity on Murray Road and the eastern section of North West Point Road.

The location of the IRPC on the north-western end of Christmas Island, far from the accommodation and port facilities in the north-east, has inevitably led to a substantial increase in traffic on the island’s roads. In particular, the traffic on the route between Drumsite and the IRPC via Murray and North West Point Roads has increased as a result of the construction of the IRPC. These two roads run through areas of high density of red crabs (Figure 25) and robber crabs. Where vehicle traffic comes into contact with crabs, it is typical that substantial numbers of crabs are killed (Hicks 1983; Morris and Adamczewska n.d.).

From January 2004 to May 2006, the BMP regularly recorded the number of live crabs, dead crabs and other dead animals, on the roads between Drumsite and the IRPC. The aim was to assess the impacts that increased traffic volumes associated with the IRPC construction would have on wildlife. See Appendix F for a description of the methods.

7.4.1 Results

A total of 811 transects counts was made in the three sections between 5 January 2004 and 26 May 2006, with counts made of the numbers of live and dead crabs, dead birds and vehicles made during the counts (Table 24). A correlation analysis showed that the numbers of dead animals increased with the number of vehicles on the road. Positive correlations were higher between vehicles and dead animals than between live and dead animals.

Table 24: Total counts of target wildlife species counted on the route from Drumsite to the IRPC in 2004–06

Section **No. of transects**	**Live red crabs**	**Dead red crab**	**Live robbers**	**Dead robbers**	**Dead birds**	**Vehicles**
Section A: Murray Road (Irvine Hill Road to Central Area Workshop) 5.3 km; 273 transects	1,803	2,550 (59%)	236	256 (52%)	115	1,375
Section B: North West Point Road (Central Area Workshop to LB4) 3.3 km; 273 transects	1,483	316 (18%)	578	86 (13%)	21	449
Section C: North West Point Road (LB4 to IRPC gate) 2.5 km; 265 transects	138	18 (12%)	128	12 (9%)	13	170
Total: 811 transects	3,424	2,884 (46%)	942	354 (27%)	149	1,994

The mortality level was much higher on Murray Road than the other sections for several reasons. Firstly, it is a longer section of road than the other two (Table 24). Secondly, there was more traffic on this section of the road. Some of this extra traffic relates to the construction of the IRPC because the Central Area Workshop (at the end of the transect section) was used for batching of concrete and other IRPC-related activities, but some of the traffic was mine-related. Other traffic (local traffic, tourism, etc) was quite low compared to the IRPC and mine traffic. Thirdly, the number of red crabs living along Murray Road in this section is very high (see Figure 25). Conversely, the low mortality of crabs on the western part of North West Point Road (Section C) reflects low numbers of live crabs in that area, following their devastation by yellow crazy ants in the late 1990s (Parks Australia unpubl. data).

Spikes in vehicles and mortality late in 2004 correspond to the relaying of underground services for the IRPC which involved considerable movements of cars and heavy vehicles. The heavy mortality in 2006 corresponds with the build-up of the final construction phase of the IRPC. In March 2006 detailed discussions were held between Parks Australia and parties involved in the IRPC construction about the extent of mortalities and options for reducing it. Subsequently, there were some but not total reductions in mortalities.

7.4.2 Discussion

There were very high, positive correlations between the number of vehicles moving on the roads and the number of dead animals counted on the roads. Increases in vehicle numbers, and concomitant increases in wildlife mortality, coincided temporally with increases in construction activity at the IRPC. There was no evidence during the sampling period of any increases in the background level of traffic not related to the construction of the IRPC.

In March 2006, Parks Australia approached parties involved with the IRPC construction, in an effort to reduce road mortality of crabs in particular. The induction seminars for the IRPC were expanded to incorporate more information about crabs, workers were educated during their 'tool box' meetings about driving with crabs, the Australian Federal Police were requested to do speed checks on the roads, and the mortality rates declined, but not to zero.

In early 2007, informal surveys showed high numbers of dead crabs on the roads to the IRPC. A total of 27 dead robber crabs was counted in a single day in February 2007 on the survey route.

7.5 Road mortality of crabs during the 2005–06 migration

Red crab migration commences at a time when crabs can meet a spawning time during the last quarter of the moon phase. The migration is triggered by the first substantial wet-season rains, and timing can vary considerably. The migration may start as early as September or October (as in 1992) or as late as January (as in 1998). Once the migration has commenced, there is some scope for predicting the timing of the consecutive events and phases because spawning peaks four to five days before a new moon (Parks Australia 2002, p. 6).

In 2005, the annual migration of red crabs on Christmas Island began on 18 October. The first rains were patchily distributed and occurred the day after the full moon, triggering a slow migration. Following a heavy downpour on the central plateau, crabs there began moving in large numbers at about midday. In other parts of the island the migration was not triggered until subsequent lunar cycles. The full extent of the migration (excluding the return of baby crabs) continued, at varying intensities over time and across different parts of the island, well into February 2006. Although this aspect was not examined quantitatively, it appears that there were three separate migrations by different parts of the population, each centred on spawning dates of approximately 30–31 November, 27–28 December and 25–26 January, respectively.

The BMP undertook road transect surveys to estimate the number of crabs killed by traffic during the 2005–2006 crab migration, and to attempt to segregate different contributions to the mortality figures (see Appendix F for a description of the methods and a map showing the location of the transects).

7.5.1 Background information about red crab migration

The onset of migration varies annually, depending on the timing of the rain within the lunar cycle. The timing of the rains leads to varying time spans between onset and spawning times. When the first rains occur approximately one week before a full moon, a fast downward migration occurs with the red crabs hurrying to meet the spawning deadline. When the first rains occur around the full moon, a slow migration occurs because the crabs have missed the next spawning deadline and have a month to make the journey. Slow migrations occur during a longer period with lower densities of crabs, while fast migrations result in high numbers of animals migrating during a shorter period of time. Depending on the timing of rain within the lunar cycle, there are intermediate situations between the two extremes. If the first rains are patchily distributed, they can trigger the migration to commence during different lunar cycles across different parts of the island (Parks Australia 2002).

During the 1981–82 migration season, Hicks (1983) investigated the road mortality of red crabs. He classified all sections of road on the island into five classes of mortality (nil, light, moderate, high and very high). He then calculated the mean number of dead crabs per unit area of road that these classes represented and extrapolated to a minimum road-mortality figure of 600,000. Allowing for undercounting he considered the true mortality figure to be between 7,000,000 and 1,000,000 crabs.

7.5.2 Mortality rates

The surveys conducted as part of the BMP suggest that an estimated 425,000 red crabs, or 1 per cent of the estimated population, were killed by traffic on Christmas Island roads during the 2005–06 migration. In contrast, the number of robber crabs killed was fairly low (107 counted, 1,200 estimated from the above assumptions).

These estimates are based on the following assumptions:

- there were probably a minimum of 40,000 dead red crabs on the surveyed roads during the surveys
- each survey covers about a week of mortality, so the three surveys represent three-sixteenths of the mortality of the 16-week migration
- approximately half the roads that cross the migration routes were counted.

This compares favourably with work by Hicks (1983), which gave estimates of between 700,000 and 1,000,000 red crabs (0.06–0.08 per cent of the population) killed by vehicles during the 1981–82 migration season. That was before any crab crossings were in place or road closures were enforced. The island’s population at that time was similar to the population in 2005 (1,200–1,500), but it is likely that there were fewer vehicles on the island then. Conversely, mining was more intensive at that time. The red crab population size is thought to be smaller now, and Hicks’ (1985) population estimate is thought to have been overly high. It is likely that the proportion of red crab population killed during the 2005–06 migration is fairly similar to the 1981–82 migration.

The number of crabs killed during the 2005–06 migration has been vastly underestimated (see Appendix F for an explanation). Dead crabs were likely underestimated in each section during each count. Surveys were conducted only on three days, but the migration lasted about four months. Crabs are killed on many other roads and the sampled route could represent about half of the high mortality zones on the island. On the basis of these assumptions, the following corrections are made.

Detailed mortality count data

The remains of about 34,000 red crabs and 107 robber crabs were counted during the three surveys on 14 sections of road (Table 25 and Figure in Appendix F). The rate of mortality (dead crabs per kilometre of road per survey) of red crabs increased with each successive count, while the rate of mortality of robber crabs remained more or less steady. The rate of mortality of red crabs was generally less where crab crossings were in use (Table 25; compare Sections G with F and H, and D with E). Conversely, in the single section that was managed by road closures (Section B), the mortality rate was very high.

No counts were made on Murray Road between Irvine Hill Road and Central Area Workshop. Red crabs were not present in any numbers on Murray Road at 8 am on 18 October, but by 12:30 pm they had started to move in large numbers. By 4 pm Murray Road was closed and traffic was diverted to the North–South and East–West Baselines. A quick estimate suggested that 2,000–3,000 red crabs were killed in the three and a half hours that Murray Road remained open. On 9 November, Murray Road was reopened, and a count of dead crabs was immediately attempted before any traffic had traversed the road, but virtually no crab remains were visible. After just three weeks a realistic count was not possible.

On 26 October, about 4,000 dead red crabs and 31 dead robber crabs were counted on the alternative route from Drumsite to the IRPC and Central Area Workshop (where the concrete batching plant of the IRPC was located) via the North–South and East–West Baselines. On 7–8 November, about 9,000 dead red crabs and 50 dead robber crabs were counted on this route.

On 5–7 December, more than 19,300 dead crabs were counted. This included almost 600 on the North–South Baseline between the East–West Baseline and the Asia Pacific Space Centre (APSC) site on the South Point section, and 3,500 on the Golf Course–Casino Road between the Mango Tree lodge and the resort. The remaining 15,000 were on the alternative route to North West Point, but unlike previous counts, this did not include the East–West Baseline between Murray Road and North West Point Road, and North West Point Road between Central Area and the East–West Baseline (LB4).

The second counts on the LB4 Link Road and Murray Road (between Central Area and the East–West Baseline) were both considerably lower than they had been two weeks earlier, and this can only be explained by a short persistence in time of the remains of dead crabs. Unfortunately, the persistence time is not known. Some sections of the road had been graded immediately before the counts (for example, parts of LB4 Link Road).

7.5.3 Attribution of impacts

Multiple users of the roads mean that precise attribution of impacts cannot be made, but some broad conclusions can be drawn.

The death of about 1,300 and 4,100 red crabs (October and November counts respectively) on North West Point Road (between Central Area and LB4) can be solely attributed to construction of the

IRPC. Prior to the November count, the road had been closed when the males began to return; but a concrete pour was in progress on the closure day and the concrete trucks continued using the road until the pour was finished. Additionally, the commuter buses continued using the road during the entire closure period, under escort by Parks Australia staff. Numerous cars and trucks followed the escorted buses, raising the mortality level even further. The failure to reconstruct the LB4 Link Road prior to the 2005 migration season (some 10 months after the final construction phase of the IRPC began) led to elevated road mortality of crabs. The transporting of concrete through a heavy migration route for crabs, in the middle of a crab migration, led to elevated road mortality of crabs.

About 1,600 (October), 2,000 (November) and 5,200 (December) deaths on the East–West Baseline and Murray Road (between the East–West Baseline and Central Area) can be mostly attributed to the IRPC traffic, although mine and other traffic contributed. The ‘Construction EMP’ for the IRPC had provisions for minimising vehicle traffic, by using commuter buses for workers and site curfews. Unfortunately, these policies were not implemented and proved inadequate in preventing high red crab mortality, particularly during the migration.

A total of 1,284 red crab deaths (November and December) on the South Point Road (North–South Baseline from the East–West Baseline to the APSC site) can be attributed specifically to mining activities, as there was very little other traffic on this dead-end section of road at the time.

About 15,600 deaths on the North–South Baseline, Drumsite and Irvine and Phosphate Hills cannot be attributed to any specific road users. The IRPC and mining apparently contributed the bulk of the traffic to these roads.

A single count in December was made on two sections of the Golf Course–Casino Road.

Between Mango Tree lodge and the golf course 730 deaths were recorded, and between the golf course and the casino 1,750 were recorded. The high number past the golf course should be avoidable as this is a no-through road and the traffic is largely leisure traffic, with the exception of access to Ma Chor Nui Nui (MCNN) Temple.

7.5.4 Effectiveness of crab management

There is considerable evidence for the effectiveness of crab management procedures (fencing and crossings) from these results. The East–West Baseline was counted in three sections: (1) east of the crossings, (2) west of the crossings and (3) between the crossings.

The section between the crossings is fenced and crabs are directed through under-road crossings. The other two sections are not fenced. The crossings and fences are strategically located where the highest numbers of crabs cross the road. The total counts for the two unfenced sections came to about 2,700 red crabs for each section, whereas the counts for the fenced section totalled about 800 red crabs. The management of crabs using drift-fences and under-road crossings reduced road mortality of red crabs by more than 80 per cent on the East–West Baseline.

Table 25: Counts of crab mortality on 14 sections of road on Christmas Island, 2005

Road section	Location description	Red crabs			Robber crabs			Length (km)	Crab management
		26/10	7-8/11	5-7/12	26/10	7-8/11	5-7/12		
B	North West Point Road (Central Area Workshop to LB4)	1,297 (393)	4,117 (1,248)		10 (3)	14 (4)		3.3	road closures
C	North West Point Road (LB4 to IRPC gate)	2 (1)	23 (9)	526 (210)	4 (2)	4 (2)	3 (1)	2.5	6 crab crossings (not used in 2005)
D	LB4 Link Road (Murray Road to NW Point Road)	80 (26)	43 (14)	12 (4)	7 (2)	1 (<1)	0 (0)	3.1	2 crab crossings
E	Murray Road (Central Area Workshop to EWB)	238 (132)	43 (24)	1,742 (968)	3 (2)	1 (1)	1 (1)	1.8	
F	East–West Baseline (westernmost grid to Murray Road)	576 (384)	1,029 (686)	1,091 (727)	0 (0)	3 (2)	2 (1)	1.5	
G	East–West Baseline (eastern to westernmost grid)	280 (175)	287 (179)	251 (157)	1 (1)	0 (0)	0 (0)	1.6	9 crab crossings
H	East–West Baseline (NSB to easternmost grid)	511 (365)	532 (380)	1,618 (1,156)	0 (0)	2 (1)	1 (<1)	1.4	
I	North–South Baseline (EWB to South Point Temple Road)		694 (107)	590(91)		9 (1)	0 (0)	6.5	
J	North–South Baseline (Lily Beach Road to EWB)	305 (40)	1,678 (218)	4,412 (573)	5 (1)	11 (1)	8 (1)	7.7	13 crab crossings
K	Phosphate Hill and Lily Beach Roads (Murray to Irvine Hill Roads)		448 (124)	400 (111)		1 (<1)	2 (1)	3.6	
L	Murray and Irvine Hill Roads (plantation gate to Lily Beach Road)	365 (140)	255 (98)	1,492 (574)	1 (<1)	0 (0)	0 (0)	2.6	Fencing
M	Murrray Road (Phosphate Hill to Irvine Hill Road)	304 (117)	1,241 (477)	3,731 (1,435)	0 (0)	3 (1)	3 (1)	2.6	
N	Casino Road (Mango Tree Lodge to Golf Course)			730 (215)			3 (1)	3.4	
O	Casino Road (Golf Course to Casino)			2,753 (765)			4 (1)	3.6	3 crab crossings
TOTALS (Figures in parenthese represent the number of crabs killed per kilometre of road)		3,958 (141)	10,390 (272)	19,348 (462)	31 (1.1)	49 (1.3)	27 (0.6)	50.5	
		33,696			107				

8. Other invertebrates

A large project to assess the potential impact of the IRPC on invertebrate assemblages in the adjacent forest was undertaken. Invertebrate samples were collected from 12 sites on a quarterly basis for two years (2004 and 2005). The data from the samples that are sorted are stored in an Access database. The unsorted samples are stored at Parks Australia.

The purpose of the study was to look at the impact on species diversity and biomass of invertebrates (excluding crabs) around the IRPC. Seasonality and differences between edge and interior forest were considered. At each sample site, trapping of invertebrates involved one canopy malaise net, one light trap, one intercept trap and four pit traps. All traps were in place for seven days except the light trap that operated for seven hours on one night.

8.1 Invertebrate collection

A reference collection of voucher specimens of invertebrates was established from the samples collected near the IRPC and augmented by incidental and adventitious collecting. The Christmas Island Arthropod Collection (CIAC) is housed in the laboratory at Parks Australia and holds a number of important specimens. These include several apparently undescribed longicorn beetles, an apparently undescribed click beetle, the second collection material of a species in the endemic genus of jewel weevils, probably several undescribed pseudoscorpions, and numerous species not recorded on Christmas Island previously. Some statistics from the collection are presented in Table 26.

The CIAC specimens and excess samples from the invertebrate collecting have been dispatched to various experts for identification and the return of identified voucher specimens. Few have been identified or returned at this stage.

Table 26: Summary statistics from the Christmas Island Arthropod Collection

Statistic	Number	Per cent
Species in collection	781	100%
Species with trap type recorded	748	96%
Taxa identified to species	156	20%
Taxa identified to subfamily	90	12%
Taxa identified to family	339	43%
Taxa not identified to order	5	0.6%
Number in vial collection	692	89%
Number in dry collection	142	18%
Number in jar collection	30	4%

8.2 Snails

The BMP worked with an independent researcher (V. Kessner) to survey the terrestrial molluscs of Christmas Island (Kessner 2006).

As reported in Kessner (2006), the survey of the terrestrial snail fauna of Christmas Island, Indian Ocean, in March 2006 revealed 38 species. This constitutes the first formal survey of the land snails by an experienced worker, the only previous survey being that by a party from the Royal Navy Survey Ship *HMS Flying Fish* in 1887. The 2006 survey recorded 38 species, of which 11 constitute presumed endemic species, 22 constitute introduced species, and five constitute species of uncertain biogeographic status (that is, cryptogenic species). Sixteen species from this total are new records for Christmas Island. One endemic species, *Georissa aff. williamsi*, is presumed extinct. Currently the number of introduced species is double that of the endemic species plus cryptogenic species combined.

Identified voucher specimens from the survey, representing 37 species, were returned to Parks Australia for incorporation into the CIAC reference collection.

8.3 Spiders

An arachnologist (V. Framenau) from the Western Australian Museum was offered the use of spider specimens from the CIAC reference collection for use in a review of the spiders of Christmas Island. A draft manuscript was sent to Parks Australia in 2006.

9. Introduced species

The BMP collected a range of data on numerous invasive animals on Christmas Island. Much of this is fragmentary, and a detailed review of invasive fauna species on Christmas Island is needed. Species need to be prioritised according to the threats they pose to native biodiversity. Control opportunities need to be identified, and in most cases developed.

9.1 Summary of results

During the reptile surveys, data were also collected on the distribution of three potentially significant invasive species: yellow crazy ants, giant centipedes and giant African land snails.

Wolf snakes, black rats and giant centipedes were identified as likely predators of the critically endangered pipistrelle. Feral cats and black rats have enormous populations throughout all terrestrial habitats on the island and have now been identified as predators of red-tailed tropicbirds *Phaethon rubricauda*.

Increasing pressures on quarantine measures mean that more invasive species are likely to colonise Christmas Island in coming years. Poor land management practices on Christmas Island have led to a proliferation of wastelands where invasive species thrive. Neglected lands surrounding the points of entry (the airport, the Port Precinct, the container depot at Drumsite, the IRPC warehouse, the Central Area Workshop, the general stores and the rubbish dump) represent particular risks.

Completion of the checklist of Christmas Island flora and fauna, particularly the process of identifying which species are native and which are introduced, is an important outstanding task.

9.2 Giant African land snail

In March 2004 a mark-recapture experiment was conducted on a dense population of giant African land snails near the IRPC. Over a two-week period 4,646 snails were marked with spray paint, using a different colour on each of three nights. After marking was complete, intensive line transect counts were made to determine abundance, movement rates and mortality rates. Preliminary analyses indicate densities of 100,000 to 200,000 individuals per hectare during the sampling period. This represents a breeding aggregation and cannot be converted to an estimate of density or biomass that extrapolates across the range. The presence of these mass-breeding aggregations is a potential weakness in the life cycle that may be able to be exploited for control purposes.

9.3 Yellow crazy ant

The exotic and invasive yellow crazy ant was introduced to Christmas Island between 1915 and 1934 (O’Dowd et al. 1999).

There is a considerable body of evidence regarding the devastating and diverse impacts that the yellow crazy ant has had on the ecosystems of Christmas Island (for example, O'Dowd et al. 1999, 2003; Green et al. 2002, 2004; Davis 2002; O'Dowd and Green 2002; Stork et al. 2003; Abbott 2004). Consequently, yellow crazy ants must be considered a prime suspect in the decline of any native species.

Yellow crazy ants have the ability to form multi-queened 'supercolonies', in which they occur at enormously high densities. In 1989, yellow crazy ants were first observed forming multi-queened 'supercolonies' and, by the mid- to late 1990s, dramatic increases in supercolony formation had been recorded (Green et al. 2004).

At their peak in 2002, supercolonies covered an estimated 2,500 hectares, which included 28 per cent of Christmas Island's remaining forest. In supercolony areas, yellow crazy ant workers occur at enormously high densities and have a significant destructive impact on the island's ecosystem (Director of National Parks 2002). Some of the most dramatic environmental impacts have been:

- depletion of the extremely abundant land crabs, which leads to changes in leaf litter build-up, seedling recruitment and weed establishment
- farming of scale which leads to weakening of trees, spread of sooty mould, and some die-back
- collapse of insect populations in the forest
- extermination of native reptiles (O'Dowd et al. 2003; Abbott 2004)
- displacement of some birds including the Emerald Dove (O'Dowd et al. 1999, Davis 2002; Abbott 2004).

An extensive control program that started in 2000 has involved aerial and hand baiting. Aerial baiting in 2002 destroyed 99.4 per cent of the yellow crazy ant population (Green et al. 2004) and alleviated a looming ecological disaster. Since that time, monitoring and hand-baiting by ground crews have continued. At the beginning of 2005, there were an estimated 300 hectares of yellow crazy ants on Christmas Island, and densities remained considerably lower than they were prior to the aerial baiting. Ant numbers are now recovering slowly and it seems unlikely that current hand-baiting techniques will be able to keep ant levels under control indefinitely. Ant densities can increase from virtually nil to serious proportions in as little as six months, so it remains possible that the yellow crazy ant population could again explode.

9.4 Giant centipede

Centipedes are known to be nocturnal, arboreal and predatory. Scolopendra centipedes in Venezuela have recently been reported catching insectivorous bats up to 15 grams with their legs, killing or immobilising them with venom, and partially consuming them (Molinari et al. 2005).

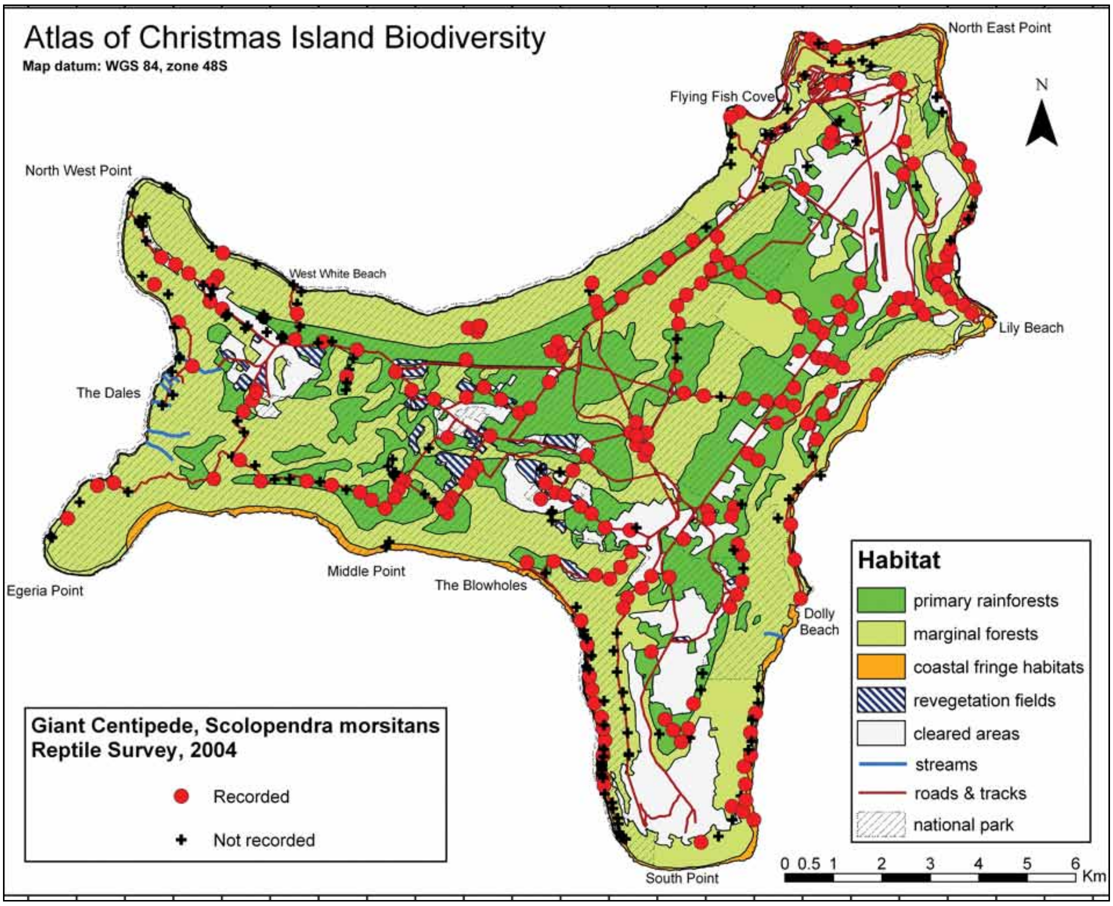
Summaries of relative abundance statistics for selected invasive invertebrate species from the quantitative reptile surveys are given in Table 27. The centipede was the most widely distributed and frequently recorded invasive pest (214 sites; 0.887 records per survey hour). A map of giant centipede distribution is shown in Figure 27.

Table 27: Relative abundance of selected invasive invertebrates on Christmas Island as recorded during reptile surveys in 2004

Measure of abundance	Giant centipede	Yellow crazy ant	African land snail
Number of sites found	214	93	13
Av. number per hour	0.887	0.354	0.049

Data from quantitative survey (320 timed surveys totalling 262:40 hours).

Figure 27: Distribution of the invasive giant centipede *Scolopendra morsitans*, as recorded during reptile surveys, February–June 2004



Centipedes are attracted to and killed by garden snail pellets (Metaldehyde at 20 g kg-1; D. James, pers. obs.) These chemicals are likely to have non-target impacts on Christmas Island. Trials should be commenced to determine whether snail pellets can be delivered on Christmas Island without serious impacts to native wildlife.

9.5 Grass skink

The grass skink *Lygosoma bowringii* was first recorded on Christmas Island in 1979, when it was initially found only in grassy habitats associated with human developments, along the developed north-eastern shore terraces, and was not found in primary or secondary forests (Cogger et al. 1983).

In 1998 it was found at 13 sites widely distributed around the island, but always from disturbed areas, and Cogger and Sadlier (2000) predicted that it would eventually be distributed throughout disturbed areas on the island.

By the time of this BMP, the grass skink was the second most commonly recorded species in the quantitative reptile surveys (149 individuals at 52 sites; 0.567 records per survey hour). It is patchily distributed, but common in some areas and has colonised several areas of remote primary forest, most notably on the upper terraces of the central north coast, and the eastern shore terrace of South Point. It is present at low density in primary forests to the north-east of the IRPC. The latter location has never been accessed by vehicles or had any infrastructure constructed.

The grass skink is also likely to colonise substantial areas of primary and secondary forest in years to come. It is not known to have any detrimental impacts. Incidental sightings since the 2004 survey indicate that the species is more widely distributed than was revealed by the survey.

9.6 Barking house gecko

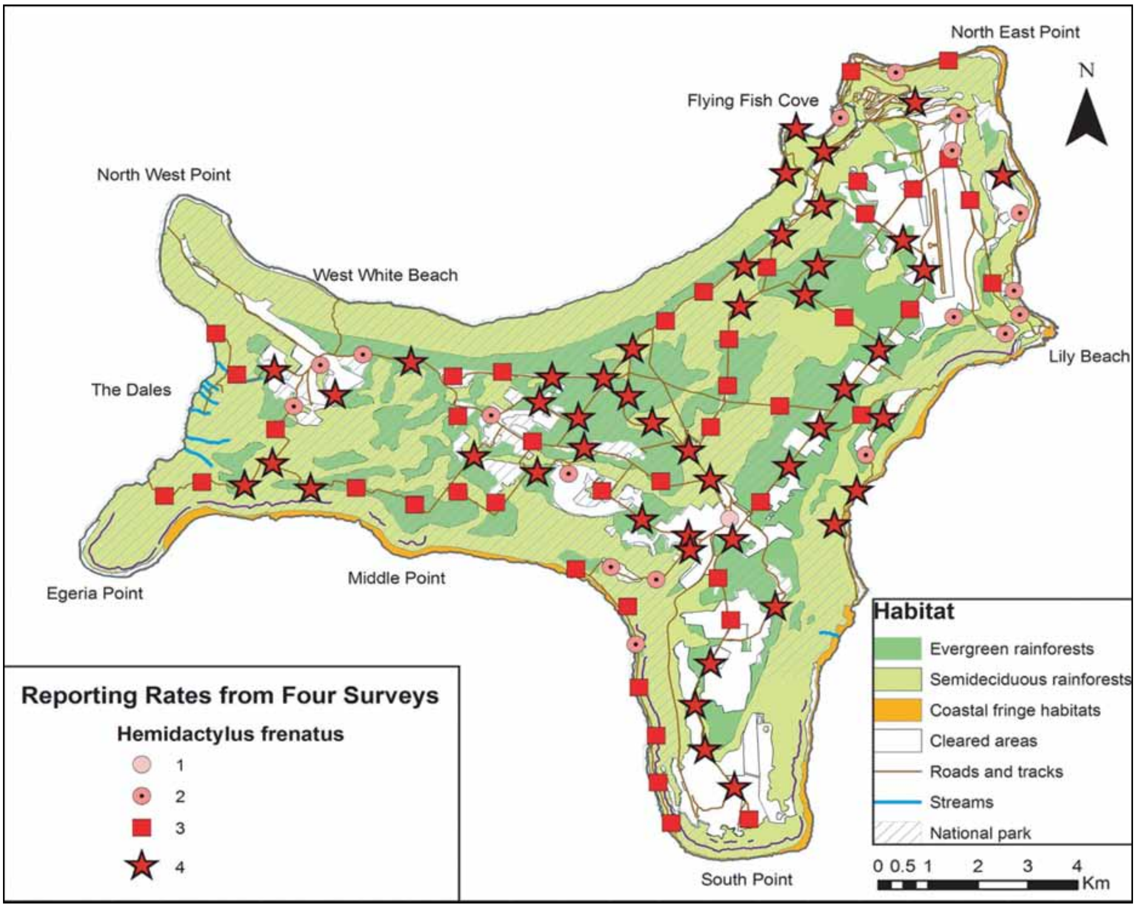
The barking house gecko *Hemidactylus frenatus* appears to have spread widely into forested areas, including primary forest near the IRPC. It might be more abundant in storm-damaged forest than pristine forest. It is also present at the IRPC construction site and the minefields surrounding the site.

It was the third most frequently recorded species in the quantitative surveys (77 individuals at 27 sites; 0.293 records per survey hour). Since this species is nocturnal and arboreal, the quantitative surveys tend to underestimate its density relative to diurnal species (skinks) and terrestrial species (giant gecko), so it is probably more common than the grass skink. It is feasible that the barking house gecko could have contributed to the decline of Lister’s gecko, but it was not possible to test this hypothesis in only three years.

In 1979, the barking house gecko was recorded at only a few sites, either associated with buildings, minefields or rocky coastal areas. In 1998 it was found in most areas surveyed throughout the island, but always in disturbed areas. In the 2004 survey, it was found widely in undisturbed coastal forests and all disturbed habitats, but only a limited number of primary forest areas. Barking house geckos are readily translocated by vehicles, in building materials, in garbage trucks, and by tree planters. It has also colonised areas where there has never been vehicle access or construction (for example, the eastern shore terrace of South Point).

The distribution and abundance of the barking house gecko were assessed by the BMP during the 2004 reptile survey, the forest bird survey, a nocturnal survey of the Christmas Island flying fox, and a brief dusk survey on the central plateau (see Figure 28).

Figure 28: Distribution of barking house gecko as recorded in the nocturnal (Christmas Island flying fox) survey, 2006



9.7 Asian house gecko

The Asian house gecko *Gehyra mutilata* was recorded at only one site during the quantitative reptile surveys (three individuals; 0.011 records per survey hour). This was in primary forest off Murray Road in the north-east of the island. During qualitative surveys, it was found to be almost as common as the barking house gecko in some buildings in the settled areas and at the Pink House, but quite rare in other buildings. The Murray Road record was the first record from forests on the island. The species has not been recorded in the western end of the island to date. Incidental observations since the 2004 survey have indicated that it is more common inside buildings than outside. It is abundant in the Kampong and at the Parks Australia plant nursery in Drumsite. Its presence at the nursery could lead to a risk of widespread translocation during tree-planting operations.

9.8 Flowerpot blind snake

The flowerpot blind snake *Ramphotyphlops braminus* is widely distributed in both primary forest and disturbed areas. It was the fifth most commonly recorded species in the quantitative survey (17 individuals at 16 sites; 0.065 records per survey hour). It was first recorded on Christmas Island in 1940 (Gibson-Hill 1950). In the 1979 survey, only one individual was recorded, but it was anecdotally reported to occur in suburban gardens at that time. In 1998, three specimens were collected from widely scattered localities. Given that this is a cryptic burrowing species, the relatively large number of records from the 2004 survey indicates that the species must be abundant and distributed virtually all over the island. It is not known to have detrimental impacts on the environment.

9.9 Wolf snake

The wolf snake *Ophites aulicus capucinus* was introduced to Christmas Island in about 1987 (Smith 1988). In the 1989 survey, the species was found only on the eastern side of the island (Cogger and Sadlier 2000). By the early 1990s it occurred at high density in the settled areas of Christmas Island (Rumpff 1992); anecdotal information indicates that the wolf snake was so common in the 1990s that they were seen on the roads in the settled areas ‘almost everytime you went for a drive at night’.

It has frequently been suggested that the wolf snake could pose a serious threat to biodiversity on Christmas Island as a predator of reptiles and bats (Smith 1988; Rumpff 1992; Fritts 1993; Cogger and Sadlier 2000; Lumsden et al. 1999, 2006). The wolf snake is known to be a specialist predator on small reptiles, particularly geckos, and occupies disturbed habitats (Karsen et al. 1986; Rumpff 1992; Fritts 1993; Cox et al. 1998). Geckos make up 48 per cent of the diet of the wolf snake in India (Wall in Fritts 1993).

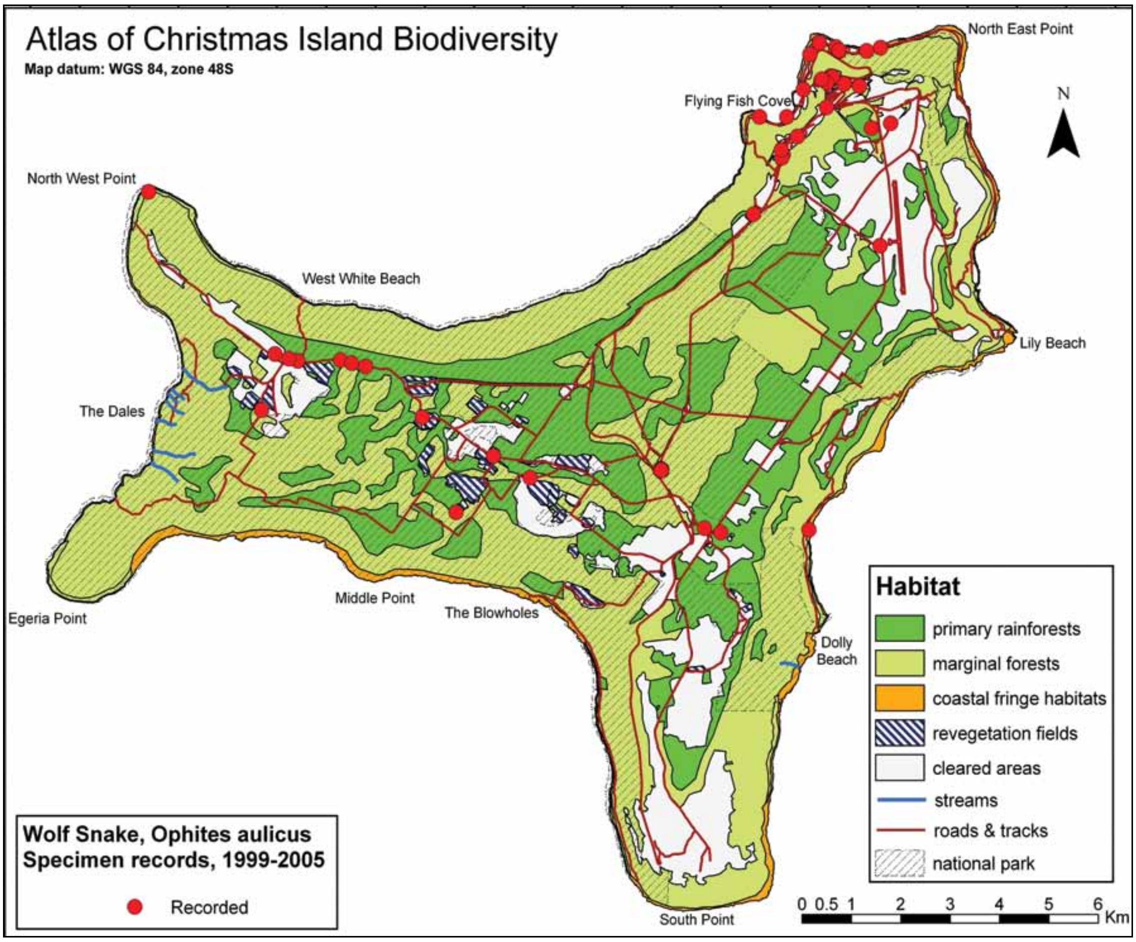
Rumpff (1992) investigated the diet of the wolf snake on Christmas Island by offering prey items to captive specimens and (separately) dissecting specimens caught in the wild. Of 200 snakes dissected, only 34 had identifiable prey remains. Geckos dominated the diet with a total of 24 recorded (Rumpff 1992). ‘House geckos’ (*H. frenatus* and/or *G. mutilata*) were offered as prey to captive snakes and the results ranged from snakes that did not eat at all to an individual that ate 27 geckos in five months. One individual was even observed eating seven geckos in two hours (Rumpff 1992).

Abundance and distribution

The wolf snake was found to have a much wider distribution than previously found. In the quantitative survey, two individuals were found at two sites (0.008 per survey hour). However, many specimens were found incidentally or brought to the Parks Australia office by members of the public. Several records came from primary forest, some a long way from roads. A number of records came from the western extremity of the island. On the other hand, the abundance of this snake and the average size of individuals both appear to have declined since the survey by Rumpff (1992). The data suggest that this snake probably occurs throughout Christmas Island, but at very low densities in most areas.

By the end of 2005, further studies of the ecology and distribution of the wolf snake (Appendix E) showed that it had expanded into the west of the island, and some records come from deep inside primary forest (Figure 29).

Figure 29: Distribution of wolf snakes from various surveys up to the end of 2005



Additional records not shown on the map include:

- one from South Point in former ML100
- several from the Circuit Track
- one from a pipistrelle roost tree in Sydneys Dale
- one more from the shore terrace at the tip of North West Point
- one from Ryans Hill
- numerous more for the IRPC and Field 26.

Indications are that wolf snakes are likely to undergo rapid population and distribution expansions in forthcoming years.

Summary statistics

Summary statistics for weights and lengths of 133 wolf snakes are given in Table 28. It is possible that the maximum size is less than it was in the early 1990s as the longest snout–vent lengths recorded in this study (nearly 600 mm) was 14 per cent less than the longest specimens reported by Rumpff (1992). A comparison of length–weight regressions for the 1992 and 2005 samples might reveal an ecological shift in the population.

Of 78 individuals that could be sexed, 44 were females and 34 were males. None of the female specimens showed evidence of eggs or young snakes. Staining techniques were not available, so egg scars and other signs could not be identified. It was therefore not possible to identify specimens that had previously had eggs or collect data on the size at sexual maturity. It was noticed that a large proportion of specimens had inflamed or burst spleens.

Table 28: Summary statistics of wolf snake measurements

Statistic	Weight (gm)	Snout–vent length (mm)	Total length
Sample size	132	100	129
Mean	15.0	352.9	419.9
Standard deviation	11.87	124.21	145.72
Minimum	1.07	154	175
Maximum	61.23	593	690

Diet

Table 29 summarises the number and proportion of reptiles from the sample of 104 digestive tracts. The heaviest total content of an individual digestive tract was 3.45 grams, which is about the weight of an adult pipistrelle. No definite traces of mammals or birds were found in the diet. Small invertebrates were recorded in 45 individuals. Reptiles made up 96 per cent of the diet by weight from the sample. Seventy-four individual reptiles from five species were recorded in the digestive tracts of 57 snakes.

Wolf snakes themselves (teeth and scales) were the most frequent species in the contents, but these remains probably do not represent dietary items, but incidental ingestion of their own teeth and scales. Cannibalism may occur. The relatively high representation of Asian house geckos (10) compared with barking house geckos (nine) reflects the fact that many wolf snake specimens came from houses and yards. The single blue-tailed skink record was from an old undated specimen, probably collected by Rumpff in the 1990s. The proportion of individuals with identifiable stomach contents was much higher than that reported by Rumpff (1992).

Table 29: Reptiles recorded in the digestive tracts of 104 wolf snakes

Species	Number of individuals and percentage contribution (including wolf snake)		Number of individuals and percentage contribution (excluding wolf snake)	
Wolf snake	30	(37.04%)		
Grass skink	24	(29.63%)	24	(47.06%)
Barking gecko	9	(11.11%)	9	(17.65%)
Asian house gecko	10	(12.35%)	10	(19.61%)
Blue-tailed skink	1	(1.23%)	1	(1.96%)
Unidentified	7	(8.64%)	7	(13.73%)
Total	74	(100.00%)	44	(100.00%)

9.10 Rats

Black rats have enormous populations throughout all terrestrial habitats on the island and, along with feral cats, have now been identified as predators of red-tailed tropicbirds.

Rats are killed by two kinds of commercial rodenticides, Warfarin and Brodificum based. These chemicals are likely to have non-target impacts on Christmas Island. Trials should be commenced to determine if these sorts of pesticide baits can be delivered on Christmas Island without serious impacts to native wildlife.

9.11 Other species

Feral beehives were found at the Chinese Cemetery and Margaret Knoll, and frequent reports were received of hives at the IRPC, where they have been an obstruction to construction activities. Feral bees are known to have had severe detrimental impacts on cavity-nesting fauna elsewhere, for example by reducing nesting habitat of parrots in Western Australia. Cavity nesting species on Christmas Island include the endemic pipistrelle, hawk-owl and golden bosun. Feral populations of tree sparrow *Passer montanus* and domestic chickens are expanding rapidly (D. James, unpubl. data). The forest bird survey identified that the nankeen kestrel is abundant in disturbed habitats across the island and has expanded in population size since the late 1980s. It has been implicated in the decline of endemic skinks (Rumpff 1992; see Section 6.1 above), and in a shift in the behaviour of the pipistrelle. The population of feral dogs *Canis lupus familiaris* has expanded from a couple to over a dozen since 2002 (D. James, pers. obs.).

A total of 53 species of ants have now been recorded from Christmas Island, and probably no more than one or two of these are native to the island (Thomas and Framenau 2006; cf. Lister 1888, p. 529 and Andrews 1900). The tropical fire ant *Solenopsis germinata* and the African big-headed ant *Pheidole megacephala* have both been established on the island for decades, are widespread, and have been associated with severe environmental impacts elsewhere in the world (Thomas 2006). Together with the yellow crazy ant, they are three of the six most serious ant pests in the world (Thomas 2006). Several species of scale insects that were identified by Abbott (2004) appear to be introduced and cause severe damage to the rainforest canopy.

Several species of flies and fruit flies are said to have colonised the island when land dumping of food waste replaced sea dumping in the late 1980s. The reptile fauna of Christmas Island is numerically dominated by introduced species, to the detriment of endemic species (see Section 6.1). The snail fauna of Christmas Island is numerically and taxonomically dominated by introduced species, to the detriment of endemic species (Kessner 2006).

The terrestrial mammal fauna of Christmas Island is restricted to four invasive species that probably caused the extinction of the three endemic species (Pickering and Norris 1996; Meek 2000).

The butterfly fauna of Christmas Island is numerically and taxonomically dominated by introduced species. By 1900, naturalists had found only nine species of butterfly on Christmas Island, excluding the three skippers. These included one endemic species (Christmas emperor *Polyura andrewsii*), two endemic subspecies (climena crow *Euploea climena* ssp. *macleari* and scalloped grass-yellow *Eurema alitha*) and one vagrant (evening brown *Melanitis leda* ssp. *bankia*). By 1940 the list had increased to 13 species, and by 1980 it stood at 22 species. With the addition of three species since then plus the inclusion of skippers, the 2006 list includes 28 species. Some of the additions since 1900 are species recorded only very rarely (i.e. vagrants), and some may have been present for much longer but were overlooked earlier. However, the vast majority are now established and they feed only on introduced plants that have become established in cleared and disturbed habitats. Whether these butterflies arrived under their own steam (that is, they blew in as adults from Asia) or came as larvae in shipments of supplies to the island is unknown.

10. Impacts of the IRPC on biodiversity

The Christmas Island Biodiversity Monitoring Program (BMP) was established to monitor the effects on biodiversity of constructing an Immigration Reception and Processing Centre (IRPC) at North West Point on Christmas Island. Construction started in July 2004. The following impacts were identified during the construction phase, and ongoing impacts are expected when the IRPC is operating.

10.1 Construction phase

The main impacts from the construction of the IRPC include:

Damage to the foraging habitat of the Christmas Island pipistrelle

The IRPC construction site is located immediately adjacent to foraging habitat used by an estimated 90 per cent of the remaining Christmas Island pipistrelles. This endemic species is critically endangered and, as noted in this report, is suffering severe decline. It is possible that, prior to the preliminary earthworks at the IRPC, pipistrelles foraged extensively on what is now the IRPC site. Clearing of secondary regrowth occurred in 2006 and 2007 including: along the Dales Road for the installation of electrical earthing cables; along the Dales Road for stockpiling of topsoil; along the Dales Road for supplying topsoil and mulch; and in Field 23 for supplying topsoil and mulch, which was known to be a foraging habitat for the pipistrelle. Clearing along the North West Point Road for the installation of utility services in 2003 and 2005 could have also impacted upon pipistrelles.

Road mortality of crabs

Extensive mortality of crabs occurred on the roads to the IRPC during the various construction phases between 2004 and 2006. In particular, the traffic on the route between Drumsite and the IRPC via Murray Road and North West Point Road increased. These two roads run through areas of high density of red crabs and robber crabs. From January 2004 to May 2006, the number of live and dead crabs was recorded on the roads between Drumsite and the IRPC. There were positive correlations between the numbers of dead crabs and the number of vehicles. Spikes in vehicle and mortality in late 2004 corresponded to the relaying of underground services for the IRPC. Heavy mortality was recorded also in 2006 corresponding with the final construction phase. Road transect surveys were used to estimate the number of crabs killed by traffic during the 2005–06 crab migration. Information that existed before the location of the IRPC was chosen to indicate that significant road mortality of crabs would eventuate from increases in traffic on Murray Road and North West Point Road. More information about survey and sampling methods is available in Appendix F.

Road mortality of birds

Along with crabs, a substantial number of native birds were killed by vehicles on the roads leading to the IRPC in 2004 to 2006.

Disturbance of forest along national park boundaries

A substantial amount of native vegetation was cleared for the construction of the IRPC along national park boundaries. This includes the boundary between the IRPC and the national park, but the most substantial clearing was along Murray Road and North West Point Road for the installation of utility services.

10.2 Ongoing operation of the IRPC

The most likely impacts of the operations of the IRPC are:

- The external lighting of the site could impact on the nesting success of Abbott's boobies and impose further impacts on the remaining foraging habitat of the Christmas Island pipistrelle.
- The landscaping process, using mulched secondary regrowth, could deliver large numbers of viable seeds of environmental weeds to the IRPC site. Weed growth may be prolific on the grounds, and the weeds could escape into the surrounding areas, much of which is national park. This process will continue for many years. For example, one weed found commonly in secondary regrowth – *Leucuaena leucocephala* – produces prolific seed, which can remain viable in the soil for a decade or more, and which is highly resilient to conventional control methods.
- There is a continuing threat to quarantine on Christmas Island from invasive species that might have already been introduced during construction, and from invasive species that can be introduced through shipment of supplies for the ongoing operation of the IRPC.
- Stormwater run-off into the national park has not been entirely addressed. Continuing problems with stormwater volumes versus keeping run-off out of the national park have led to the redesign of the stormwater systems several times. However, drains leading to the boundaries with the national park continue to reappear whenever stormwater loads peak.
- Wildlife mortality on the roads to the IRPC is expected to continue with high traffic loadings due to the operation of the IRPC.

10.3 Recommendations for the IRPC

The recommendations to mitigate the environmental impacts of the IRPC are listed below.

Monitoring program for IRPC operations

A program should be established to monitor the potential impacts of the operation of the IRPC.

Quarantine to be strengthened

Christmas Island's ecosystems are highly vulnerable to threats posed by invasive species. Managing invasive species has consumed considerable resources on the island in recent years, and the expense of management is only likely to increase. The ongoing operation of the IRPC should enhance protection of biodiversity by preventing invasive species from arriving on Christmas Island.

External lighting

Monitor the impact of external lighting on Abbott's boobies and the Christmas Island pipistrelle and adopt measures to minimise impacts such as reducing night-time lighting hours and intensity if required.

Weed control

Control the growth of weeds within the IRPC to prevent spread into the national park and surrounding area.

Stormwater run-off into the national park

Reassess the site stormwater drainage system to prevent run-off into the national park.

Wildlife mortality on roads

Wildlife mortality on the route to the IRPC during the operational phase should be monitored and could be minimised by transporting staff by bus to and from the centre via the North–South and East–West Baselines.

11. Conclusions

The biodiversity of Christmas Island has international significance, with 253 endemic taxa, 165 species of national conservation significance, and 110 species listed as protected in the schedules to the EPBC Act (Table 30). Overall, 576 species were classed as significant. These statistics are unlikely to be matched by any other small island or national park in Australia.

Nearly 70 of these species are missing (not recorded in decades despite dedicated surveys), most of which are endemic and the status of many other species is unknown. Some are extinct, some are probably extinct. Many other species are now known to be in severe decline. Five of the six native reptiles are on the verge of extinction. The three endemic terrestrial mammals are all but gone. The two bats are both in rapid decline. And many endemic invertebrates have not been found for over 50 years. During 110 years of settlement the biodiversity of Christmas Island has been facing accelerating pressure. Initially, impacts were slow to emerge, but since the 1980s many species have begun to decline. It is more than likely that declines, disappearances and extinctions will accelerate exponentially.

The BMP attempted to investigate the causes of declines of several threatened species, some covered by recovery plans and some not officially recognised as threatened. Given the large number of threatened and declining species on Christmas Island, it is not feasible to investigate the proximal threats on a one-by-one basis and then develop specific mitigation measures for each case. Attention needs to focus on addressing the root causes of biodiversity loss on Christmas Island. These root causes can be summarised as follows:

- **Invasive species management:** Invasive species management begins with quarantine procedures, but must also include early detection systems (routine monitoring) and early response procedures (rapid control of newly arrived invasive species). Effective control of invasive species that have already colonised Christmas Island will be crucial to ensuring the long-term protection of the island’s biodiversity.
- **Land management:** Poor land management practices are widespread and diverse on Christmas Island. Only a small percentage of minefields have been adequately revegetated. Neglected wastelands develop into areas ideal for supporting weeds and invasive animals. Existing roadworks and weed control practices frequently facilitate the expansion of weed populations. On the whole, areas of unmanaged wastelands are expanding rapidly. Although considerable effort and resources have already gone into weed management and yellow crazy ant control, there is more to do and this need will continue for some time.
- **Global factors such as climate change:** Climate change (and other global issues) might pose a serious threat to the biodiversity of Christmas Island, but there is limited scope to address these issues on the island itself. In the meantime, the effects of poor invasive species and land management practices are already having serious impacts and need to be addressed urgently.

Table 30: Summary of status of 576 significant animals and plants on Christmas Island

Category	Status level	Vertebrates	Invertebrates	Plants	All
EPBC listings	A1. Extinct	2	0	0	2
	A2. Critically endangered	1	0	2	3
	A3. Endangered	5	0	1	6
	A4. Vulnerable	7	0	0	7
	A5. Migratory	92	0	n/a	92
	A6. Marine	63	0	n/a	63
	Subtotal[2]	**107**	**0**	**3**	**110**
Significance	B1. Internationally significant (endemic)	24	205	24	253
	B2. Nationally significant (only/virtually only population in Australian jurisdiction)	38	34	93	165
	B3. Locally significant	25	0	11	38
	Other	113	4	0	117
	Subtotal	**200**	**243**	**128**	**576**
Population estimate	Extinct	2	0	0	2
	Missing[1]	3	57	7	67
	Rare	28	72	32	132
	Unknown	59	74	33	166
	Uncommon	15	14	14	43
	Common	15	14	14	43
	Abundant	5	8	28	41
	Visitor	19	n/a	n/a	19
	Vagrant	59	n/a	n/a	59
	Other	0	4	0	4
	Subtotal	**205**	**243**	**128**	**576**
	Grand total[2]	**205**	**243**	**128**	**576**

[1] Species not recorded in more than 50 years (invertebrates) or more than 20 years (vertebrates and plants).

[2] The subtotals and the grand total do not sum evenly because the inclusion criteria are not always mutually exclusive.

11.1 Future work priorities

Christmas Island pipistrelle

The Christmas Island pipistrelle is declining at a rapid rate. Despite some progress, recent research has failed to identify the precise cause or causes of the decline. Ongoing research and management are high priorities for the Director of National Parks; actions to be considered include:

- monitoring communal roost sites and foraging areas
- protecting roost trees from predators
- assessing the effectiveness of artificial roost boxes
- identifying potential roost trees.

A captive breeding program for the pipistrelle is also being investigated.

Christmas Island flying fox

A greater understanding of the Christmas Island flying fox and its conservation is needed. Actions for consideration include:

- studying the population ecology and interactions with the forest ecosystem
- identifying threats so that appropriate management actions can be implemented
- research into roost sites.

Land crabs

A collaborative approach between land managers is needed to manage impacts on land crabs. In particular, future planning decisions on Christmas Island regarding the location of infrastructure and the subsequent traffic demands these place on roads are needed to ensure that environmental impacts on crabs are minimised. The most important issue in managing crab mortality on roads is to keep crabs and vehicles apart. Actions for consideration include:

- installing more crab crossings on the North–South and East–West Baselines
- greater resources to implement existing crab management initiatives and implement technological improvements, such as over-road crossings and permanent drift-fencing
- modifying concrete kerbing on roadsides so that they do not trap crabs on the roads
- preparing road users for road closures during migrations and reducing traffic volumes on Murray Road
- developing a more systematic approach to counts of road mortality during migrations to assist in assessing management actions.

Invasive species

Invasive species control should be prioritised according to the threats they pose to biodiversity. The Director of National Parks will continue to implement invasive species control management in the national park, focusing on weeds and yellow crazy ants. A collaborative approach with other land managers is needed outside the national park to ensure efforts are effective. These efforts will require substantial ongoing funding.

Nominations for species listing under the EPBC Act

Consider recommending new species listing under the EPBC Act or changing the category of listings of some currently listed species.

Ongoing research into biodiversity on Christmas Island

The BMP has enabled a greater understanding of the species and ecosystems on Christmas Island to be developed, together with the threats to their conservation. The results from the program have provided a foundation for expanding scientific knowledge, including building on the species databases. Funding is required to continue the research and surveys to build this information.

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Appendix A: Pipistrelle research and monitoring

Equipment: ultrasonic bat detectors

Most of the mapping and monitoring components of this study involved the use of ultrasonic bat detectors to record the location and abundance of pipistrelles.

Three sets of AnaBat II Bat Detector® units connected to AnaBat CF Storage ZCAIM® units (supplied by Titley Electronics, Ballina, NSW) were used to detect and record the ultrasonic echolocation calls made by flying (foraging or commuting) bats.

The detectors reduce the frequency of the bat calls to an audible level and the ZCAIM units record the calls along with the date and time to a compact flash card and allow the programming of automatic on-and-off times. CF storage ZCAIM Interface© software was used to download data files from the flash cards to computer and convert them to sonograms, which could then be viewed and analysed using Analook© 3.1b software.

For some of the mapping surveys, the bat detector system was operated from a moving vehicle. In these circumstances, Garmin GPS Map76® hand-held global positioning system receivers were plugged into the ZCAIM units and the latter would then record the GPS position every two seconds to the compact flash card. The GPS locations were then converted into shape files using Esri® ArcMapTM 9.1 GIS software to show transect surveys overlaid with the locations of bats that were recorded.

For monitoring and mapping at stationary points, the detector set-ups were mounted in waterproof boxes and placed on top of custom-designed tripods.

The transducer microphones, which are very sensitive to moisture, were faced downwards under the protection of the box. Calls were reflected up to the transducer from a plate mounted directly beneath it. In 2004, galvanised steel plates mounted horizontally were used. From mid-2005 onwards, plexiglass plates mounted diagonally beneath the transducer were used to prevent distortion of call rebounds between the reflector plate and the tripod head. The detectors could thus be left on station for extended periods without risk of damage from weather or animals.

Detectors at stations were set to automatically switch on at 6 pm and off at 6 am. The number of pipistrelle calls recorded by the detectors per night provides an index of the abundance of the species. It does not provide information on how many bats there are (absolute abundance), but can provide a reliable index for comparing differences in abundance from place to place and over time (relative abundance).

Transect surveys

Lumsden et al. (1999) conducted transect surveys in 1998 to map the distribution of pipistrelles across the island. This established a baseline data set for drive transects. The total transect lengths were approximately 240 kilometres in 1998.

The BMP conducted a second set of transect surveys between February and April 2004 and again in July 2005. Bat detectors (with GPS coupling) were operated from the tray of a vehicle travelling more slowly than 20 kilometres per hour, and covered most of the island's roads and vehicle tracks. The 2004–2005 surveys were designed to be as comparable as possible to the 1998 surveys. The main purpose of the surveys conducted as part of the BMP was to identify areas where pipistrelles forage. However, the data can also be compared to similar surveys conducted in 1998 by Lumsden et al. (1999).

- Between February and April 2004, approximately 180 kilometres of transects were driven around the island with a bat detector. Surveys were conducted on eight nights for a total of approximately 32 survey hours. The fieldwork was curtailed by heavy rains before South Point could be surveyed.
- Between June and July 2005, approximately 80 kilometres of transects were driven with a bat detector. Surveys were conducted on four nights for a total of approximately 16 survey hours.

Stationary detectors

Lumsden et al. (1999) provided a quantitative baseline on relative abundance of pipistrelles in 1998, by recording bat calls at 84 stationary detector sites across the island. (These sites are called 'Lumsden' sites in this report.) In 2004, 43 'Lumsden' sites were resurveyed and, in 2005, 35 of them were resurveyed. Altogether 51 'Lumsden' sites were surveyed at least once in 2004 or 2005. Survey effort was less in the 2004 and 2005 surveys, but effort was concentrated at sites where bats had been recorded in 1998.

This facilitated a comparison of the number of bat calls recorded at each station over the two sampling periods. Lumsden et al. (1999) counted the number of calls (equivalent to the number of times a bat passes in range of the detector set-up) between 6:30 pm and 9 pm. This is a relative index of abundance that allows comparisons between sites or over time but does not provide a population estimate. In 1998, technology allowed the recording of pipistrelles for only two and a half to three hours, so sampling was restricted to between 6:30 pm and 9 pm. Advancements in technology allowed sampling in 2004 and 2005 to extend from 6 pm to 6 am. To compare the 2004 data set with the 1998 study, the number of calls in 2004 recorded at stations between 6 pm and 9 pm were counted and these counts were corrected for the additional half-hour of recording time. Each site was scored on a four-point rank scale of frequency of bat calls. However, the scale changed slightly over the years to optimise the advantages gained from technological advances which increased potential sampling periods. The scales used were a compromise between keeping results comparable between years and using the benefits of the improved technology. Changes in the scale were conservative in that it generally became easier to reach higher ranks (for example, a rank of high was more easily achieved in 2006 than in 2004). Furthermore, about 85 per cent of calls occur before 9 pm, so the difference in the number of calls recorded over 12 hours *versus* three hours is much smaller than the difference that the sampling periods might suggest. Thus the changes generally led to slight overestimates of increases and slight underestimates of decreases between years.

Bat detector set-ups on tripods were placed overnight at stationary sites around much of the island between April and July 2004 and again between September and October 2005. Sites included many of the same sites used in 1994 and 1998 by Lumsden and Cherry (1997) and Lumsden et al. (1999) respectively, as well as a range of additional sites to increase the sampling effort in some portions of the island. The majority of sites was surveyed for only one night in each year. This facilitated both an assessment of the contemporary distribution and relative abundance of the pipistrelle, as well as an assessment of trends in distribution and abundance between 1994, 1998, 2004 and 2005. The number of interior sites that could be surveyed was restricted by the difficult terrain, and the number of exterior sites was restricted by the amount of that habitat.

Late in 2005, a finer scale was adopted for mapping and monitoring pipistrelles, following results indicating that the population was confined to the western end of the island. Bat detector set-ups were stationed at points on a 500-metre grid across the western 15–20 per cent of the island in order to define areas where the pipistrelles feed. The points on the 500-metre grid were selected *a priori* and were mostly situated away from roads inside the forest. This provided an opportunity to investigate an issue identified in the National Recovery Plan (Schulz and Lumsden 2004): ‘Action 5: Identify primary foraging sites away from ecotones and roads’.

At the same time and in the same general area, the detectors were also placed along tracks and around the edges of clearings in Mining Leases 138, 139 and 140 (Fields 25 and 26), and around the Immigration Reception and Processing Centre (IRPC) construction site. This gave a further 33 ‘exterior’ sites along roads, tracks, in minefields and around forest edges. An additional 14 exterior sites were surveyed in Field 26 in January 2006.

In 2006, 12 sites were selected for more frequent fine-scale monitoring. These sites were in four groups of three: three in the centre of the densest feeding area at ML140, three around the periphery of the feeding area at ML140, three at an apparent, low-density outlying site along the North West Point Road, and three at another apparent, low-density outlying site in Field 22. The three bat detector set-ups were stationed at sites for approximately one-week intervals and rotated through the sites systematically. Thus it was intended that each of the sites would be sampled for approximately seven nights each month. These four clusters of sites sampled all of the known pipistrelle foraging areas as identified during the mapping studies of the previous two years.

Table A1: Definitions of scores for ranking the relative abundance of pipistrelles from bat-detector data between 1998 and 2006

Rank	Definitions		
	1998	2004–2005	2006
High	≥150 calls from 6:30 pm to 9pm (≥1 call/min)	≥180 calls from 6:30 pm to 9pm (≥1 call/min)	≥180 calls from 6:30pm to 6am (≥1 call/min)
Medium	15 to 149 from 6:30 pm to 9pm (≥1 call/10 min, < 1 call/min)	18 to 179 from 6:30 pm to 9pm (≥1 call/10 min, < 1 call/min)	18 to 179 from 6:30pm to 6am (≥1 call/10 min, < 1 call/min)
Low	1 to 15 calls per night (sample times varied)	1 to 18 calls per night (from 6 pm to 6 am)	1 to 18 calls per night (from 6 pm to 6 am)
None	0 calls per night (sample times varied)	0 calls per night (from 6 pm to 6 am)	0 calls per night (from 6 pm to 6 am)

Mapping and monitoring of roost sites

In December 2005, researchers from ARI trapped 52 pipistrelles at sites in Field 26 (Lumsden et al. 2006). Attempts to catch bats at Field 22 were unsuccessful, apparently because of the low density of bats there. Twenty-four individuals were tracked with micro-radio transmitters during their study, and seven maternity roost sites were found (Lumsden et al. 2006).

In January and March the BMP undertook limited visual monitoring of pipistrelles emerging from some of the roost sites. This involved being positioned under the roost trees at dusk and using binoculars, a night vision monocular (Night Owl Optics® NOCX3M with 3.1 x magnification) and a bat detector to visually count pipistrelles as they emerged from the roost sites for the night. In April 2006, bat detector set-ups were placed under the roost trees to monitor emergence. The temporal patterns of calls recorded by the detectors can indicate whether or not the roost sites are being used, but do not provide a count of bats using the sites.

In April 2006, the BMP installed four automatic remote cameras at pipistrelle roost trees to capture images of potential predators that might be accessing the roost sites. Fauna Focus FF120 camera set-ups (supplied by FaunaTech in Bairnsdale, Victoria) were custom-designed prototypes developed with funding from the Natural Heritage Trust grant. Digital cameras mounted on tripods in waterproof housings, and using infrared illumination in dark conditions, were automatically triggered by motion sensors. The sensors were active infrared beams mounted to partly encircle the trunk of the roost trees, in such a way that anything moving up or down about two-thirds of the circumference of trunk would break the beam. Because the known colonial roost sites in 2006 were all under loose flaking bark on dead trees, the cameras were installed at the bases of the trees for occupational health and safety reasons. The cameras are thus capable of continuously monitoring activity on the trunks for extended periods, in the order of months.

By mid-2006, only three of seven communal roost trees located in December 2005 were still standing, and one of these was no longer being used by the pipistrelles. In August 2006, Glenn Hoye (a bat biologist from Sydney, NSW) visited the island for a week and assisted the BMP in trapping and radio-tracking pipistrelles to locate more roost trees (Hoye 2006). Two additional trees were found and remote camera set-ups were moved to them in September 2006. Bat detectors were also placed under some of the roost trees in April, May, November and December 2006.

Installing artificial roost boxes

Seven sets of artificial roost boxes were installed near known roosting locations in the west of the island, in May 2006. Each set was mounted on a six-metre high steel pole and steadied with guy-ropes. Each set contained two boxes and four separate cavities, theoretically sufficient to house more than 100 pipistrelles. Given the rate at which roost trees had fallen over in 2006, one purpose of installing the boxes was to determine if a shortage of suitable roost sites may be a cause of the decline, whilst simultaneously helping to mitigate that potential threat. Likewise, being on steel poles and isolated from the forest canopy, the artificial sites would be more difficult for most potential predators to access, so they could mitigate that potential problem while simultaneously investigating whether such mitigation is beneficial. Some species of insectivorous bats readily adapt to artificial roost sites, while others do not. Amongst those species that do colonise boxes, some do so quickly and others do so slowly. The pipistrelle’s reaction to artificial roost boxes is unknown. Monitoring of roost sites is necessary to determine their effectiveness.

Assessing potential threats

Four cameras were deployed at five roost sites and two control sites, for a total of 663 trap nights between April and December 2006. The camera took 6,163 photographs on 328 trap nights, but did not capture any predation events. A total of 99 individual animals was photographed on roost trees, and ants (*Campenotus* spp.) were recorded swarming on tree trunks on 11 nights. No animals were recorded on either of the control trees. The cameras were moved as other data indicated which roost sites were active.

During 2005, Ruth Marr undertook a study of the diet of the wolf snake *Ophites aulicus capucinus* to determine if the snakes had the potential to prey on pipistrelles. Apart from specimens collected by Parks Australia staff, members of the general public on Christmas Island were also invited to supply specimens. Over 100 specimens were dissected and their gut contents identified. Length, weight, gape size, sex and reproductive status of each snake were recorded. The collection locations of most snakes were also mapped on GIS. The results of this research are summarised in this report as they have significant bearing on the pipistrelle program.

Appendix B: Forest bird survey methods

Between February 2005 and January 2006 the BMP conducted a survey to develop a baseline index of relative abundance for forest birds across Christmas Island. The vocalisations of the Christmas Island flying fox enabled it to be surveyed at the same time.

The survey involved recording whether each of eight birds and two other vertebrate species, the Asian house gecko and the Christmas Island flying fox, were detected or not detected during 10-minute surveys. Both visual and auditory cues were used to determine presence. The survey site was effectively the radius around the stationary observer in which each species could be detected. The diurnal surveys were conducted along roads and vehicle tracks between 6 am and 12 pm.

Three factors were considered in designing the study:

1. resource efficiency
2. ease of repeatability to facilitate follow-up surveys
3. statistical power to detect future trends.

The adopted approach was to record the presence or absence of the target species during 10-minute counts at fixed points across the island, and to repeat the surveys four times to generate statistical power. A total of 128 survey sites was selected across Christmas Island. Sites were spaced at 500-metre intervals along roads and vehicle tracks. At each site the habitat type was recorded in four broad categories. The island was divided into six sectors to analyse geographical effects. Counts were also classified by time of day and by season. Stationary surveys for presence or absence of each target species were conducted at each site during a 10-minute sampling period, using both visual and auditory evidence. Surveys were repeated four times (n = 128 x 4 – 1 = 527 counts).

The 128 sites were stratified by habitat, and each site was surveyed four times (once in February, May, September 2005 and January 2006). Four replications of the survey were conducted between 1 June and 12 July 2006. Each survey was completed in 7–13 days. Surveys were undertaken only between 6 am and 12 noon. Approximately 12 sites could be surveyed per day. Efforts were made to randomise the time of sampling periods at each site between surveys, but this proved to be difficult to achieve due to geographical and logistical constraints. Cloud cover, wind strength and precipitation were also recorded, but have not been analysed.

There is an acknowledged bias inherent in counting only in areas accessible by vehicles, but it was considered necessary to make the survey efficient and therefore more easily repeated in the future. Presence/absence data are binomial (i.e. the data can take only one of two values). At each site a species could be recorded between zero and four times generating reporting rates of 0 per cent, 25 per cent, 50 per cent, 75 per cent or 100 per cent. When pooled, the data give total reporting rates in increments of approximately 0.25 per cent. Sites were classified by habitat and by geographical sectors of the island, whilst surveys were classified by time of day and month. In this way reporting rates were compared for different habitats, sectors, times of day and months, using Chi square homogeneity tests and comparison of 95 per cent confidence intervals were used to identify specific differences.

For the Christmas Island flying fox, nocturnal surveys were carried out between 6 pm and 1 am, and 17 sites were omitted due to access difficulties. The diurnal and nocturnal forest surveys for *s* were designed to set baseline indices of relative abundance that can be repeated in the future to detect upward or downward trends in abundance. While the diurnal survey measured the relative abundance of animals roosting away from camps, the nocturnal survey measured the relative abundance of animals feeding. These surveys will have the most statistical power for detecting changes when reporting rates are close to 50 per cent and will have symmetrical decay of power around the 50 per cent mark. The low reporting rates in the diurnal survey (4.7 per cent reporting rate) give the survey insufficient statistical power to detect small and probably medium-sized trends. The statistical power of the nocturnal survey (>18 per cent reporting rate) is better, although its power has not been assessed to identify the minimum detectable change.

Christmas Island goshawk

The BMP funded a month-long banding expedition of the Christmas Island goshawk by the Australasian Raptor Association in August 2004 (see also Hurley 2005). This was the first survey of the species, and no baseline data on the population previously existed.

The species is uncommon and difficult to detect, so suitable survey methods are necessarily detailed. Actions 2 and 3 of the National Recovery Plan for the Christmas Island goshawk (Hill 2004) prescribed that a trapping and colour-banding project that allowed individual identification of banded birds would be a suitable method for assessing population size and structure and developing a long-term monitoring program. It also recommended that the best way to do this would be to fund an expedition of intensive banding by volunteers from the Australasian Raptor Association.

As a result, 56 Christmas Island goshawks were colour-banded, including 34 males and 22 females. The age structure of the sample was 27 juveniles (one-year-olds), 10 two-year-olds, and 18 adults (three years and over). Subsequently, a resighting program was established by the BMP. Staff from Parks Australia, members of the Christmas Island public, and visiting scientists and naturalists contributed to the resighting program, recording the age, sex, colour-band combinations (if any) of the bird, as well as time, date, location and habitat details. Almost 300 sight records were recorded between August 2004 and August 2006, and colour-banded birds were recorded 69 times.

In 2006 Natural Heritage Trust funding was secured to continue the banding program. A second expedition was mounted in August and September 2006 and a third expedition was underway in March 2007 while this report was being prepared. By October 2006, 103 birds had been banded with individually recognisable colour combinations. The banding and resighting database contained 440 Christmas Island goshawk records, of which 103 were banding records, 99 were resightings of banded birds, and 238 were sightings of unbanded birds.

Appendix C: Flying fox survey methods

A survey of the Christmas Island flying fox was conducted from December 2005 to January 2006. It aimed to repeat most of the methodology utilised by previous surveys from the literature to make results obtained as comparable as possible. In addition to a limited literature review, this included:

- ground aural/visual searches of areas in the vicinity of the previously known camps
  ground aural/visual searches of possible areas of flying fox abundance (for example, where bats were heard in the island-wide survey/fruit bat survey, areas mentioned by members of the public)
- road aural/visual transects conducted both during the day and at dusk
- dusk observations from five vantage points
- observations of feeding areas, both at dusk and during the evening with the aid of night vision equipment/spotlights
- boat transects of the north and west coasts
- incidental observations.

The ground aural and visual searches are fairly self-explanatory and were conducted with two observers. Some areas (such as the area around Middle Point) were visited more than once and at different times of day. The road transects were variously conducted on bicycle at night, by car by day (with one observer placed in the tray of the vehicle) and by car at dusk, relying on visual observations. All these road transects were visited at least twice during this survey. At camps, trees were marked with a plastic tag (inscribed with the letter F and a number), bats counted, the species identified and GPS coordinates obtained. These trees could then easily be rechecked on subsequent visits. Complete counts were made to reduce error, such as that seen in the results of Entwistle and Corp (1997) where patch counts were employed.

The vantage points included were Margaret Knoll, Stronach Knoll, Ryan’s Hill, South Point and a cleared area known as ‘Helicopter Hill’. Margaret Knoll, Stronach Knoll and Helicopter Hill had been previously used for such work (Tidemann 1985, Corbett et al. 2003). Numbers of bats seen were gathered in 15-minute segments as per Tidemann (1985). Additional information, such as direction of travel, was noted at most locations, but this information will not be presented here.

Observations of feeding areas were done similarly, with counts of individual bats and the number of trees being utilised at 15-minute intervals. Unfortunately, the data obtained in this survey were mostly confined to an area of flowering coconut trees known as ‘Cocy Corner’, and so are not directly comparable to similar observations conducted at Ross Hill Gardens by Tidemann (1985). This was due to there being no fruiting *Muntingia calabura* trees during the time of this survey. With the aid of night vision equipment, ‘flying visits’ were made to all feeding sights in single evenings to ascertain an approximate number of bats using all such areas.

Appendix D: Seabirds survey methods

Abbott’s booby

Surveys were conducted in June to August 2004, and August to September 2005 and 2006. A grid of 300-metre plots was established in Abbott’s booby nesting habitat adjacent to the northern (NNE) side of the IRPC and adjacent to the northern side of the North West Point Road, east of the IRPC. The plots were selected and marked out *a priori* by GIS, but had to be modified after groundtruthing, because of inaccessible terrain. There were five experimental plots adjacent to the boundary of the IRPC (plots A to E), and three control plots along North West Point Road (Plots G to I). There was also a small triangle between the experimental and control plots (plot F). Unfortunately plots C, D and E could not be surveyed effectively because of the difficult terrain and vegetation conditions, and so they were abandoned.

Surveys were undertaken by a team of two to four people, walking abreast in transects perpendicular to the forest edge. String lines were used to ensure even and systematic coverage. Nest trees were tagged with orange plastic tags (about 15 x 75 mm) engraved with unique numbers, nailed to the nest tree. The survey teams changed with each survey, and observer bias was not quantified. Nests and potential nests were assessed and categorised using the proforma of the 1991 survey (Yorkston and Green 1992).

This survey had severe methodological limitations. Abbott’s boobies probably first breed at six to eight years and the generation length (equivalent to the average age of nesting birds) is probably closer to 20 years than 10 years. The breeding cycle takes 15–18 months so successful pairs nest only once every two years, at most. Unsuccessful pairs might nest in successive years. Some pairs might take rest years. Therefore, only a proportion of the breeding population actually breeds in any given year, and that proportion is dependent on the nesting success of the previous year. Consequently, measurable change in the population may take decades to unfold, depending on the statistical power of the survey design and on how the term ‘measurable’ is defined. In this survey, statistical power was weak because of:

- the short duration of the study compared with the life cycle of the species
- the small sample size (limited by the number of available plots)
- unmeasured and uncontrolled differences in survey effort between years
- potential confounding effects such as the effects of yellow crazy ants, forest change and canopy density which could not be incorporated into the design.

Appendix E: Reptile survey methods

Reptile studies by the BMP began with a wide-ranging survey in 2004, followed by a number of more specific studies of selected species and groups.

In the first half of 2004, an extensive survey was undertaken to map the distribution of native and introduced reptiles across the island.

Quantitative searches involved two to three people searching an area of forest by raking the leaf litter, rolling and splitting fallen logs, and stripping bark from fallen and standing trees. Searches were timed (average 20 minutes) and generally covered about 0.5 hectares. Carefully standardising and recording the effort allow calculations of *catch per unit effort* figures, so that the status of each reptile species can be quantified and compared. Qualitative surveys of various types were undertaken, including spotlighting surveys, miscellaneous observations, and searches for single species. Whilst these surveys are not quantified, the records are included in the GIS maps of reptile species to give greater resolution of reptile distribution across the island.

The survey was conducted between 17 January and 23 June 2004. A total of 409 sites was surveyed, including 320 quantitative searches and 89 qualitative searches. A total observer effort of 262 hours 40 minutes was put into the quantitative surveys.

Monitoring endemic skinks at North West Point

The study was undertaken from July to November 2004. The study area was located on the coastal terrace of North West Point (560953E, 8845496N). The habitat at the site was semi-deciduous vine thicket dominated by trees including *Gyrocarpus americanus, Celtis timorensis, Kleinhovia hospita, Terminalia catappa* and *Pisonia grandis,* with an understorey of *Pandanus christmasensis*, *Carmona retusa, Colubrina pendunculata, Maclura cochinchinensis* and weedy herbs. The forest floor was littered with rocks and logs. Leaf litter was light to moderate. Crab densities were low following ant invasions. Canopy cover varied, producing a mosaic of light patches on the forest floor and tree trunks. Light penetration increased steadily from July to October but was not quantified.

On 5 July a pilot study was used to develop methodologies and establish 12 plots for counting skink numbers. The plots were arranged in a split-plot, factorial design. Two subsites A and B (about 500 metres apart along the coastline) each had six plots in a grid, three at the top of the cliff edge and three 40 metres inland of the cliff edge. The plots were selected for their suitability and were not random. The plots were marked by flagging tape at a central point, from which skinks were counted by eye. The census area of each plot differed depending on the density of vegetation (which affects observer visibility), microhabitat (surface area for foraging skinks) and amount of sunlight penetration. It was not feasible to control these variables across the plots.

A 10-minute skink count and two 30-second yellow crazy ant counts were made in each plot during each sample period. The observer crept into the centre of the plot and waited motionless for two minutes. Then a 10-by-10-centimetre card was placed on the ground and the number of ants crossing the card in 30 seconds was recorded. A second ant count was then made in the same way. If yellow crazy ants were seen but not recorded in the counts, this was noted. A 10-minute count of skinks was then made while the observer remained as still as possible. Skinks were counted in a 360 degree circle to a radius of as far as the observer could see with the naked eye (generally about 10 metres). Efforts were made to keep track of the locations of individuals so that they were not counted more than once. Ambient conditions (particularly sunshine and time of day) are likely to affect activity levels of skinks. Therefore, all counts were undertaken between 9 am and 2 pm on days that were not heavily overcast. Time of day and average cloud cover (sunny, patchy, overcast) were recorded. Five different observers participated during the course of the experiment, but no effort was made to assess observer biases.

The 12 plots were surveyed four times before ant-baiting (7 and 15 July, and 6 and 15 August). The Invasive Species Team baited around the study site on 18 August. Presto 01TM was distributed by a hand crew at a rate of three kilograms per hectare. The 12 plots were then surveyed four times after baiting (13 and 29 September, 13 October and 10 November).

There was no opportunity to establish a control site where baiting did not occur, as there was only one population found on North West Point with only enough room for one plot in the study area and no other skink populations could be found nearby. A control site could have been established on South Point, but the distant location and different habitat would have rendered it of dubious value as a control site.

In addition, to assess the effects of ant baiting on non-target forest arthropods, insect samples were collected before baiting (7 July) and after baiting (13 October). On each occasion, a single sweep sample was collected in each plot. A sweep sample consisted of 10 sweeping movements brushing around vegetation, with a butterfly net. These samples were sorted and matched against the Christmas Island Arthropod Collection (see Section 8.1).

The skink-count and ant-count data were analysed in three-way ANOVA models, with treatment (pre and post baiting) and distance from the cliff (cliff top and interior) as crossed factors, and plot as a nested factor. The subsample sites were not incorporated in the model. Sample occasions were treated as replicates, four before and four after baiting. The dependent variable for the skink model was the count of skinks in each plot. Ant counts were combined as the sum of the two counts plus one (if ants were not recorded in the plot) or plus two (if ants were recorded either in the counts or incidentally). Cloud cover was run as a covariate on a nominal scale of one to three. Alpha probability level was set *a priori* at 0.05.

Monitoring endemic skinks at South Point

Two intern projects were supervised by the BMP to investigate the ecology of the blue-tailed and forest skinks on the western shore terrace of South Point, along the Boulder Track (Gorge 2005; Rueff and Bordelet 2006). The location was chosen because the largest populations of both species occur there together and the access is reasonably good.

A study site was established, based on methods used in the blue-tailed skink monitoring study. Fifteen sites were selected in the study area, with a layout of five columns and three rows. One row was along the outer fringe of the vegetation, coastline, another was along the centre of the terrace and the third was along the base of the inland cliff. The columns were 40 metres apart. The coastal row and the inland row were 40 metres apart, but there was no fixed distance between the inland row and the cliff row, because of the relief of the cliff. The first plot of the coastal row was randomly selected; the other plots were not random. The plots were marked by a central post. The census area of each plot differed depending on the density of vegetation and the amount of sunlight penetration. However, the density of the understorey vegetation was low and the maximal distance of sight was limited by the ocular capability of the observer (approximately 15 metres). An inner circle with a radius of four metres was marked out with flagging tape around each post to assess densities.

Wolf snake survey methods

The distribution, abundance and diet of the wolf snake were investigated by Ruth Marr for the BMP in 2005. Transect surveys to assess density were undertaken at night, around the settled areas in an attempt to replicate Rumpff’s (1992) baseline survey. An advertisement was placed in the local newsletter requesting people to report sightings and hand in specimens to augment records from other Parks Australia surveys. A total of 132 snakes was measured and weighed and 104 individuals were dissected. Reproductive organs were examined and dissected to sex individuals, and assess breeding condition. The digestive tracts were examined to identify dietary items.

Appendix F: Crab survey and sampling methods

Red crab population surveys

Red crabs were sexed and measured between 12 February and 25 May 2004. The survey period was after the migration when crabs had returned to the forest. Samples were collected from forested locations on the plateau and middle terraces (Central Area, the Plantation and Ross Hill Gardens) and the coastal and lower terraces (Greta Beach, The Blowholes, the Chinese Cemetery and Ethel Beach).

Selecting crabs randomly (or at least without bias) was desirable, but proved to be a challenge. When crabs are approached, they scurry into hiding, and increasing human activity in an area increases the level of the crab’s avoidance behaviour. Sampling was mostly undertaken during very wet weather when the crabs’ avoidance reactions appeared to be lower. A team of two or three people would select an area with high to moderate crab densities. Crabs were collected immediately upon selecting a site and over a short period of time (about 5–10 minutes). They were caught by hand and held in plastic tubs until processing. Collectors were instructed to catch the crabs nearest to them at all times, regardless of size. After a few minutes when most of the crabs in the surround had either fled or been captured, logs and bark were turned over to search for hiding crabs. This last activity was designed to locate smaller crabs, which can hide more easily, but collectors were still instructed to collect crabs based on proximity rather than size.

Crabs were then sexed and measured. Sex was determined by the shape of the abdominal shield (broad on females, narrow on males). They were measured across the widest point of the carapace with Vernier callipers, to the nearest 0.5 millimetre.

Samples of baby crabs were measured across the widest point of the carapace as they emerged onto the shore at different points around the island during the return in March 2005. A sample of approximately 200 baby crabs was collected just after they emerged from the sea onto Greta Beach on 24 February 2005. They were raised in a terrarium until August 2006, when the survivors were preserved and measured. In the terrarium the crabs were fed on leaf litter, occasionally supplemented with road-killed crab and fish-food flakes or pellets. The soil was kept moist to provide suitable humidity conditions.

Robber crab population surveys

Robber crabs were sexed and measured between June and September 2004. The survey period was after the migration when crabs had returned to the forest. Samples were collected by driving around the island’s roads and vehicle tracks by day, searching for robber crabs. There are three acknowledged biases in this approach: robber crabs are mostly nocturnal; only individuals on or beside tracks were sampled; and young robber crabs occupy ‘hermit’ shells and so can be hard to find.

When crabs were located they were captured by hand, sexed, measured and released. Sex was determined by the presence (females) or absence (males) of pleopods on the left-hand side of the abdomen (Hicks et al. 1990). They were measured across the widest and longest point of the carapace with Vernier callipers, to the nearest one millimetre, and weighed with an electronic balance to the nearest one gram. Any damage to limbs, claws or carapace was recorded.

Road mortality surveys

In the early stages of the BMP, the field team regularly drove between Drumsite and the IRPC construction site. From early January 2005 we began to systematically count the number of dead animals along the route. It was presumed that the numbers of dead animals (predominantly crabs) on the roads would be related to the numbers of live crabs and the numbers of vehicles on the roads. We therefore included counts of live crabs and vehicles in the transects. The route was divided into three sections as defined in Table . Counts were made outside the red crab migration periods, which were counted separately.

Counts were made by driving slowly and visually inspecting the road surface. Road verges were excluded from the counts. The numbers of each target species were counted visually and recorded in a notebook before being transferred to a spreadsheet database. Tally counters were used when high numbers of the targets were encountered. Transect counts were assumed to take place instantaneously, although they varied in length from five to 20 minutes per section depending on road conditions and the amount of wildlife counted. During times of high crab mortality, crab carcasses were spray painted with road-marking paint to avoid double-counting on subsequent transects.

Table F1: Descriptions of road sections with transect counts of dead wildlife, 2004–06

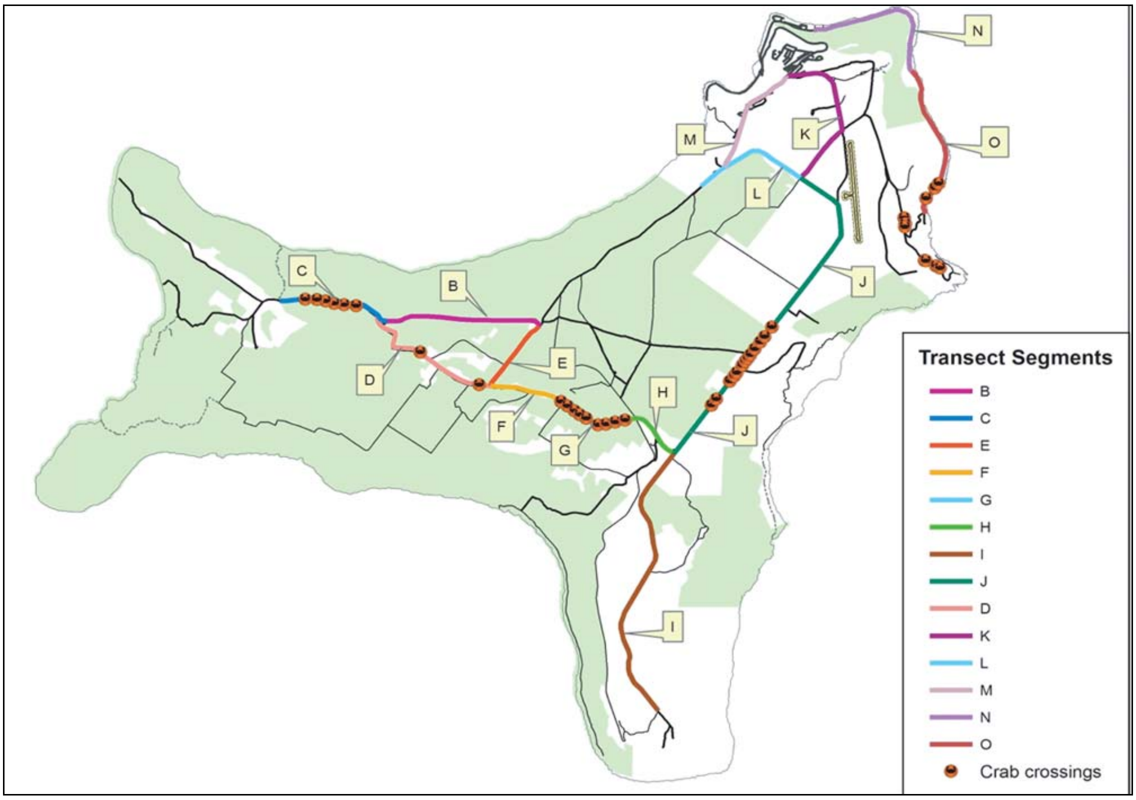
Section	Description and length of road section monitored (km)		Management
A	Murray Road (Irvine Hill Road to Central Area Workshop)	5.3 km	Road closures
B	North West Point Road (Central Area Workshop to LB4)	3.3 km	Road closures
C	North West Point Road (LB4 to IRPC gate)	2.5 km	6 crab crossings (not operated in 2004–06)

Annual migration surveys

Counts of red crabs and robber crabs killed along 14 sections of the roads were made on 26 October, 7–8 November and 5–7 December 2005 (Table 25 and Figure). Roads were driven slowly and dead crabs were counted by the driver and/or a passenger, using tally counters. Counts were made by David James in October and November, and by Kent Retallick and Kim Orchard in December.

Not all of the sections were counted on each occasion because of issues such as road closures and construction activities. On each occasion the route between the settled areas and the IRPC construction site was covered as well as possible (excluding Murray Road between the Dryers and Central Area Workshop, which was closed). Additional sections were added in later counts to facilitate comparison of impacts between IRPC traffic and other (background) traffic.

Figure F1: Location of road sections surveyed for road-killed crabs during the 2005–06 migration period



It is unlikely that every dead red crab on the road was counted during a survey because of the large numbers of crabs involved. In some cases the density of dead red crabs was so high that identifying individuals was not possible in an efficient time period, and in such cases a conscious estimate was made to underestimate rather than to overestimate. The remains of dead red crabs have only a short persistence time on the road surface (sources of loss of remains probably include: general decay; washing by heavy rains, consumption by other crabs; scattering by the traverses of other crabs; pulverisation by vehicle traffic; and covering with dust and mud). The impact of these factors likely varies between road sections and surfaces.

In many cases, a single dead red crab leaves more than one mark on the road. The most common cause of this is a crab stuck to a tyre, which leaves a straight line of evenly spaced marks. In these instances it is clear that only one crab is involved. In other cases a crab may be cut in half and the two halves eventually squashed far enough apart that there appears to be two crabs involved. Efforts were made to reduce this bias by not counting small or partial carcasses.

As a consequence, any estimates of road mortality are almost certainly an underestimate. Overall, underestimation was considerably higher than overestimation, and it must be assumed that the counts represent only a small proportion of the true road mortality.