ECOLOGY OF THE HONEY POSSUM, *TARSIPES ROSTRATUS*, IN SCOTT NATIONAL PARK, WESTERN AUSTRALIA.

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The Honey possum, *Tarsipes rostratus,* is an obligate nectarivore, known to feed on plant species from only three Families in south-western Western Australia: Myrtaceae, Proteaceae and Epacridaceae. These plants can be adversely affected by fire, decreased rainfall or groundwater levels and the pathogen *Phytophthora cinnamomi*. We investigated the ecology of *T. rostratus* in terms of: (i) how the population fluctuated in response to rainfall and fire over a 20-year period and (ii) changes in diet and movements during a period of decreased food availability in late summer. Mean capture rates were significantly positively correlated with mean flowering rates of *Banksia ilicifolia* over a 20-year period. Winter capture rates were also significantly positively correlated with both annual and winter rainfall two years prior to trapping in recently burnt areas, but not in long unburnt areas. Capture rates were significantly higher in unburnt *Banksia* woodland during winter but densities there have declined since 1996, associated with the death of many *Banksia ilicifolia* trees from persistent *Phytophthora* infection. Notwithstanding this decline, winter capture rates in the unburnt areas were still approximately double those in the burnt areas 6 years after the last fire. Short-term capture rates were negatively correlated with barometric pressure, showing that movement and foraging is stimulated by the passage of low pressure frontal weather systems. Despite the paucity of known food sources flowering in late summer and autumn, there was no evidence of *T. rostratus* using plant species from other than the three above-noted Families. Utilisation areas in summer were also no larger than those previously recorded across all seasons in Scott National Park. Some individuals, however, moved extensive distances to locate spatially restricted food sources. The conservatism of their diet and the sensitivity of the population to changes in rainfall and fire history indicate that *T. rostratus* populations are particularly vulnerable to some of the environmental changes now occurring in south-western Australia.

Keywords: *Banksia*, fire, home-range, marsupial, nectarivore, *Phytophthora,* rainfall, *Tarsipes rostratus*,

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THE Honey possum, *Tarsipes rostratus*, is the only non-flying mammal to live on an exclusive diet of nectar and pollen and is the sole representative of the Family Tarsipedidae (Kirsch 1977; Kirsch, Lapointe and Springer 1997). *T. rostratus* is a predominantly nocturnal/crepuscular inhabitant of the south-west of Western Australia, occurring in floristically diverse heathlands (kwongan) dominated by the Families Proteaceae and Myrtaceae (Russell and Renfree 1989). Their diet is thought to be restricted to plants belonging to the Families Myrtaceae, Proteaceae and Epacridaceae (Wooller *et al*. 1984). In the areas investigated thus far, *Banksia* is their primary food source (Weins, *et al.* 1979; Wooller *et al.* 1983; Wooller *et al.* 1993) and the abundance of *T. rostratus* fluctuates in concert with *Banksia* flower production (Wooller *et al.* 1993).

There are several threats to their food plants that may negatively impact upon the ability of the ecosystem to support local populations of *T. rostratus*. These plants can exhibit a decline in distribution and vigour through decreased water levels (Groom *et al*. 2000b; Groom *et al*. 2001) and the pathogen *Phytophthora cinnamomi* (Sage *et al*. 2004; Shearer *et al*. 2004). Of major concern is the significant decline in rainfall recorded in the southwest of Western Australia over the last decade with current streamflow into dams being only 57% of that recorded in the period 1975-1996 (Water Corporation of WA 2006). Similarly, a fire régime inappropriate to the life history of kwongan vegetation can lead to local extinction of some species (Bell *et al*. 1984; Meney *et al*.,1994). The combination of these factors makes local populations of *T. rostratus* susceptible to decline in the short term, and potentially places the species at risk in the long term.

T. rostratus consume a large volume of nectar relative to their body size (6-12 g), with a daily intake of approximately 7 mL of nectar and 1 g of pollen required to maintain mass balance in a 9 g individual (Bradshaw & Bradshaw 1999, 2001). Consequently, during periods of decreased nectar supply, they show poorer condition as measured by weight and tail fat (Wooller *et al*. 1981). Despite high food requirements and a specialised diet, early studies of movements and utilisation areas of *T. rostratus* suggested that the species has very small home ranges. Home ranges calculated solely from trapping records in Fitzgerald River National Park, for example, were only 0.13 ha for males and 0.07 ha for females (Garavanta *et al.* 2000). Direct measurement of movements by radio-telemetry within Scott National Park, however, has shown that *T. rostratus* are capable of moving greater distances, with utilisation areas of males averaging 0.79 ha and females 0.14 ha (Bradshaw & Bradshaw 2002), with individual movements up to a distance of 500 m during a single night.

As an obligate nectarivore with a restricted suite of food plants, *T. rostratus* may be particularly susceptible to factors that impact negatively on these plant species. Therefore, our investigation focused on (i) how their abundance fluctuates in response to changes in rainfall and fire régimes, and (ii) whether individuals respond to a paucity of suitable food by increasing their home ranges or by consumption of a wider than normal range of plant species. Wooller *et al.* (1998) showed that abundance was correlated with rainfall from the preceding year in Fitzgerald River National Park. The present study, however, has been carried out in Scott National Park, some 500 km west of Fitzgerald River

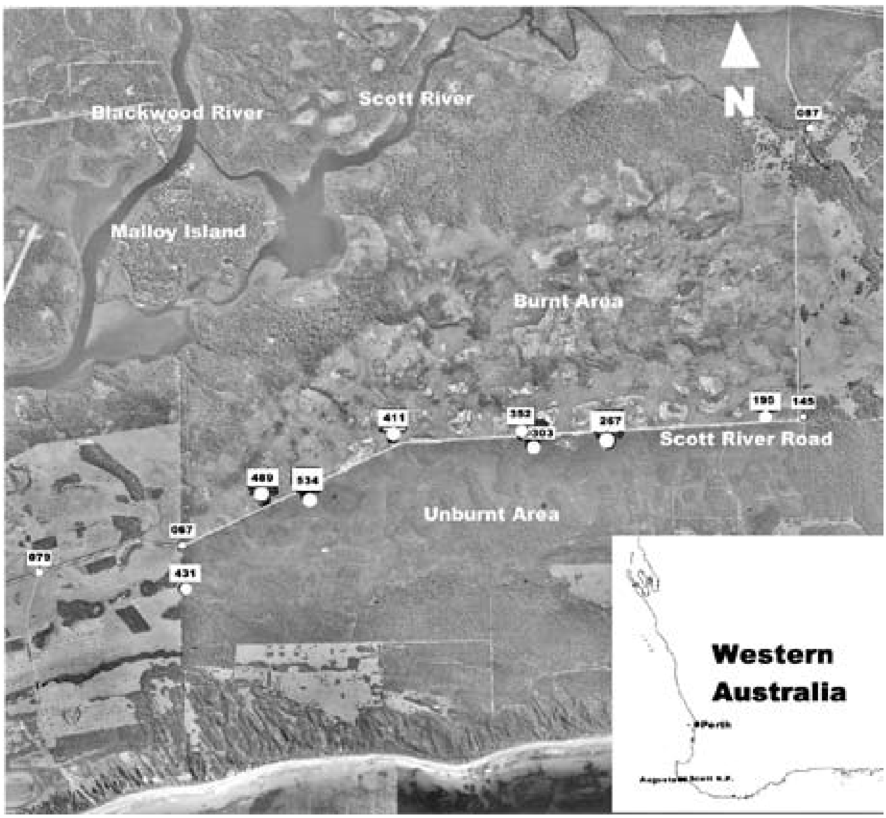

Fig. 1. Aerial view of the Scott National Park study area, extending for approximately 6.5 km on either side of Scott River Road. Sites of traplines are indicated by clusters of waypoints with 489 = Site 4, 534 = Site 7, 411 = Site 6, 352 = Site 9, 303 = Site 5, 267 = Site 1 and 195 = Site 8. Closed early in the study was 431 = Site 3. The portion of the study area burnt in 1993, and again in 1999, lies immediately north of Scott River Road; the southern portion appears not to have been burnt for at least 40-50 years.

Trapsite	Habitat	Number of Traps	Configuration	Distance Apart (m)
1	Unburnt *Banksia ilicifolia* woodland	73	25, 30, 18	8
3	Unburnt jarrah, *B. grandis* woodland	30	3 x 10	5
4	Burnt *B. ilicifolia* woodland	60	3 x 20	5
5	Unburnt *B. ilicifolia* woodland	36	3 x 12	5
6	Burnt *B ilicifolia* woodland	60	5 x 12	5
7	Unburnt *B. ilicifolia, B.meisneri* woodland	45	3 x 15	5
8	Burnt *Beaufortia sparsa s*wamp/ sedge/heath	50	5 x 10	3
9	Burnt *Beaufortia Sparsa* swamp/Sedge/heath	50	5 x 10	3

Table 1. Details of trapsites established in Scott National Park study site over the period 1987-2006.

National Park, and in an area that is floristically less diverse, with fewer species of *Banksia*. The results of a 20-year trapping program were used to test for fluctuations in abundance with rainfall and fire. A short-term study utilising radio-tracking and identification of food species was undertaken during late summer of 2004 to investigate potential changes in foraging behaviour associated with low food abundance.

MATERIALS & METHODS

Study Area

The study was undertaken in Scott National Park (34°17.0’S, 115°13.8’E) in the extreme southwest of Western Australia. The vegetation is predominantly low swamp/sedge/heath, punctuated by deep sandy ridges supporting sparse woodlands dominated by *B. ilicifolia* growing to 5-6 m. These woodlands have an understorey similar to the heath in Fitzgerald River National Park (see Wooller *et al.,* 1981). The swamp/sedge/heath habitat, however, is not duplicated in the Fitzgerald River Reserve. The study area is bisected by Scott River Road and the section to the north was burnt in November 1993 and April 1999 by the Department of Conservation and Environment (DEC, formerly CALM), when fighting wildfires. The section of the study area south of Scott River Road has not been burnt for at least 40 years and the impact of fire in the northern area is evident in the aerial photograph shown as Fig. 1. The distance between waypoints 067 (Johnson Road) and 145 (Milyeannup Road), representing the western and eastern limits of the study site respectively, is 5.6 km. The major food plants for *T. rostratus* occur

Year	Sites Trapped	No of Trap Nights	Density (Inidivs. ha^{-1})	Capture Rate (%)
1987	1,3	213	2.09	4.41
1988	1,3,4,5	875	1.67	1.49
1989	1,3,4,5	1200	5.26	1.71
1991	1,4,6	1018	3.03	0.59
1992	1,3,4,5,6	1833	2.70	1.88
1993	1,3,4,5,6	2399	4.79	3.14
1994	4,6	406	0	0
1995	1,4,5,6,7	3083	3.42	2.45
1996	1,3,4,5,6,7	1316	12.17	8.73
1997	1,4,5,6,7	1577	8.36	7.53
1998	1,4,6,7	973	9.44	6.88
1999	1,4,5,6,7	1207	10.75	8.59
2000	1,4,5,6,7	1673	4.14	2.55
2001	1,4,6,7	723	5.29	4.55
2002	1,4,5,6,7	2491	17.45	6.76
2003	1,4,5,6,7,8,9	3217	12.92	2.65
2004	1,4,5,6,7,8,9	5782	3.14	1.22
2005	1,4,5,6,7	1375	3.73	3.04
2006	1,4,5,6,7,8,9	2391	5.79	1.65

Table 2. Schedule of trapping for *Tarsipes rostratus* in Scott National Park from 1987-2006.

commonly in the woodland areas. *B. ilicifolia*, *B. meisneri* (to 1.5 m in unburnt Site 7) and *A. obovatus* (to 1 m) flower in winter and spring while *A. meisneri* (to 1 m) flowers in spring and summer. The flowering phenology of these species is shown schematically in Fig. 2. *Beaufortia sparsa* (to 2m) is widespread in the swamp/sedge/heath habitat, while *Banksia occidentalis* (to 3 m) has been found in only two localised patches of swamp/sedge/heath in the extreme eastern portion of the study area. Both these species flower in autumn. *Corymbia calophylla* (marri) occurs as isolated trees to 5 m, mainly in the southern unburnt section of the study

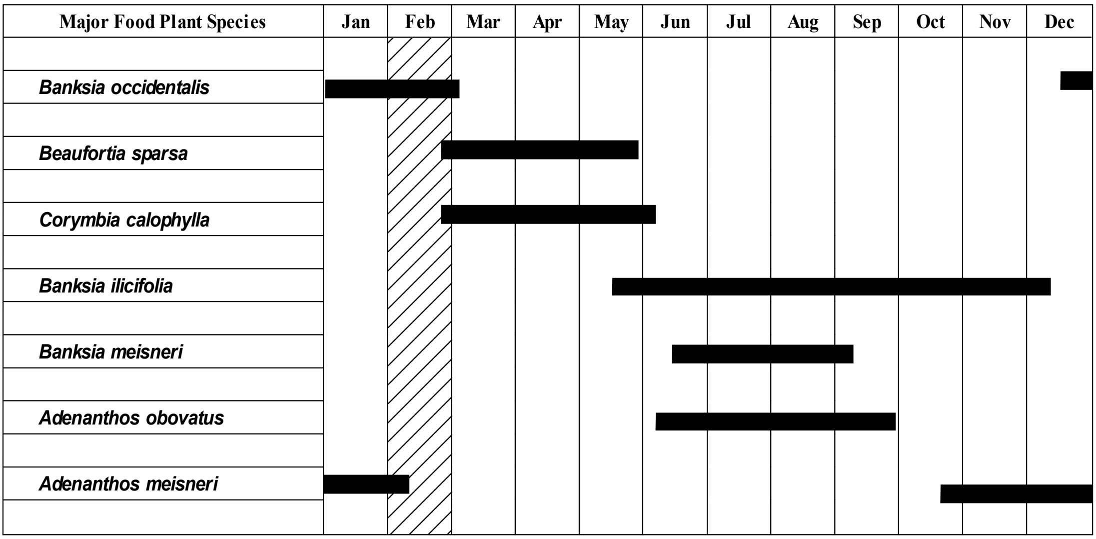

Fig.2. Flowering phenology of the seven major food plants of *Tarsipes rostratus* in Scott National Park: *Banksia occidentalis, Beaufortia sparsa, Corymbia callophylla, Banksia ilicifolia, B. meisneri, Adenanthos obovatus* and *A. meisneri.* The hatched month of February highlights a period of the year when flowering of food plants is most limited, with only *Banksia occidentalis* flowering for the whole period.

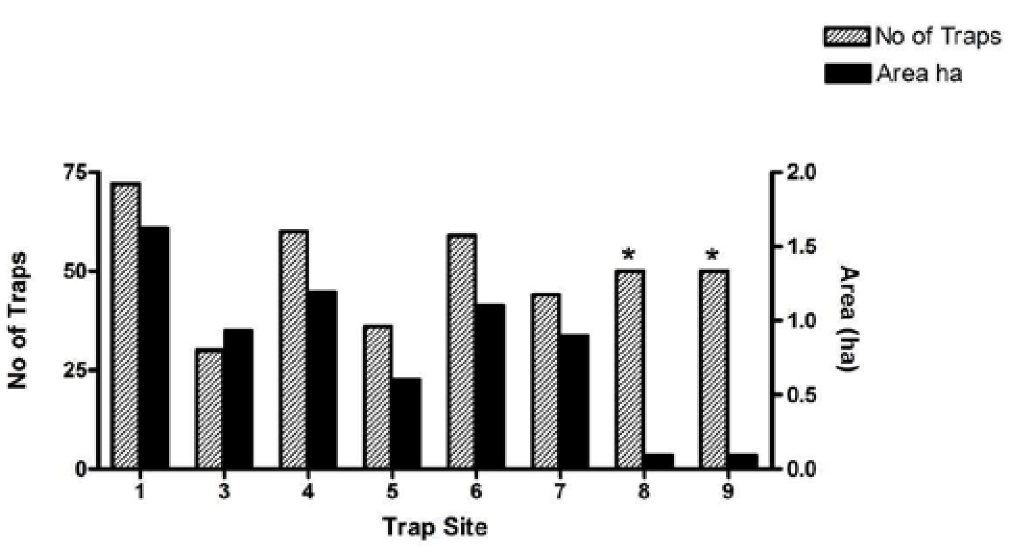

Fig.3. Details of the main trapping sites in the Scott National Park study area, showing the total number of traps per site, plus the immediate area delineated by the traplines. * Traplines 8 and 9 were established in 2003 in the swamp/sedge/heath area and opened only in late summer and autumn when *Beaufortia sparsa* was in flower.

area, and also flowers in autumn. *Eucalyptus marginata* (jarrah), growing to 3 m as a mallee, is also found in isolated clumps throughout the study area and flowers in spring.

Capture of Animals

Honey possums were captured in pitfall traps constructed of 500 mm lengths of 150 mm diameter PVC drainage pipe that were sited primarily in the isolated *Banksia* woodlands. Traps were arranged from 3 – 8 m apart in lines ranging from 12 to a maximum of 30 traps per line, at eight different locations, as shown in Fig. 1 and Fig. 3, and were closed with metal lids when not in use. Details of the various traplines are given in Table 1 and Fig. 3. also shows the area in ha of the section of bushland delineated by the traplines at each site. Trap densities were much greater in Sites 8 and 9, with traps laid 3 m apart in a rectilinear grid. These two traplines were established in *Beaufortia sparsa* swamp/sedge/heath late in the study, in 2003. Trapsite 3 was established in *Eucalyptus marginata/Banksia grandis* habitat but was relatively unproductive and little used during the long-term study. Table 2 summarises the trapping schedule for the period 1987-2006, giving details of sites trapped and capture effort. A short study was undertaken in the summer 2004 to identify diet and document animal movements during a period of anticipated low food supply for *T. rostratus*. Traps (see Table 2) were opened for twelve consecutive nights between 13-24 February 2004 and checked and cleared between 06:30 and 08:30. Captured animals were placed individually in calico bags and returned to a nearby field station to be sexed, weighed (± 0.1 g), marked individually (ear punch, World Precision Instruments) and brushed for pollen grains. Animals were released at their site of capture after processing and all procedures were approved by the Animal Ethics Committee of the University of WA.

Long-Term Records

Long-term trapping data acquired by SDB and FJB across 34 trapping sessions in Scott National Park between 1987 and 2006 were analysed. Seasonal variation in capture rates in each trapping site, expressed as animals caught per 100 trap-nights (%), was analysed using ANOVA and Tukey's *post-hoc* test to establish the source of the variation. The sites were then compared statistically for significant variation, prior to the first fire. Post-fire data from burnt sites (Sites 4 and 6) and the unburnt sites (Sites 1, 5 and 7) were pooled where appropriate to analyse the impact of fire on capture rates. The impact of rainfall on capture rates was investigated by regression analysis using monthly rainfall records for Scott River Station (34°18'1"S, 115°23'34"E) from 1998 to 2006 provided by the Bureau of Meteorology (W.A.). Regressions were plotted between *T. rostratus* capture rates and total annual rainfall for one and two years prior to capture, and against winter rainfall (June-August) for one and two years prior to capture. Annual changes in density, recorded as individuals captured per ha, were analysed by ANOVA across sites in order to detect any long-term trends in population size and animal abundance. Rainfall and barometric pressure were recorded daily from January 2004 with an Environdata WeatherMaster 2000 automatic weather station (Environdata Australia Pty Ltd, Queensland). The correlation between daily capture rates and barometric pressure measured at 0900 hr during the passage of two low-pressure frontal systems was recorded in September 2005 and October 2006. The impact of fire on capture rates was tested using a paired Mann-Whitney U-test to compare the mean capture rates between burnt and unburnt *Banksia* woodland. Statistical analyses were conducted using Instat® (Graphpad) and the Statistica® package (Statsoft 1995, 2000).

Identification of Pollen Grains and Diet

Pollen grains were collected by wiping animals' snouts and head with fuchsin-stained agar blocks (Beattie, 1971) and by staining faeces collected from the calico bags. Pollen grains were identified using a micrometer graticule under light microscopy by comparison with reference slides prepared for all flowering species in the area. The assumption that pollen grains on the snout reflect actual consumption was confirmed by comparing faecal and snout samples from each individual.

Sugar Concentration in Nectar

Nectar was collected by inserting glass microcapillary tubes into the corolla of *A. meisneri* and by manual centrifugation of *B. sparsa* (Armstrong and Paton, 1990) and stored frozen. The sugar concentration was measured with a temperature-compensated hand-held Brix® refractometer.

Blossom Counts

B. ilicifolia was identified early in the study as the primary food supply of *T. rostratus* in the Scott National Park study area and counts of fresh inflorescences were made during each trapping period. Six trees were marked individually in each trapping area and inflorescences were recorded during each trapping session with the aid of field glasses. The reliability of counts made by different observers was tested statistically before acceptance. Trees that died of *Phytophthora* infection were replaced periodically with uninfected trees from the same site.

Radio Tracking

We outfitted six males with Holohil BD-2T temperature-sensitive radio-transmitters (Holohil Systems Inc., Canada) in February 2004. Transmitters were glued to the skin using Vetbond® tissue adhesive to a small shaved patch at the back of the neck. The bias towards males resulted from their considerably higher trapping rate, as has been observed in previous studies (Wooller *et al*. 1981; Wooller *et al*. 1993). Pre-calibrated radio-transmitters weighed approximately 0.7 g, constituting less than 10 % of a possum's body mass. The effective range of the transmitters varies from over 200 m in open regions to 20 - 30 m in dense vegetation (Bradshaw and Bradshaw 2002). We used both Biotrak Model 3 (Sirtrak, New Zealand) and Australis 26 K (Titley Electronics) telemetry receivers and 3-element Yagi antennae for tracking possums.

Animals were released at the site of capture between late morning and early afternoon on the day of capture. Tracking tagged animals commenced between 20:00 and 23:30 hours on the evening of release. Animals were located each morning, midday and night until the end of the study period or when the radio-transmitter fell off. Each position was marked and logged using GPS (Magellan and Garmin eTrex®).

Calculation of Utilisation Areas

Home range is usually defined as the area traversed by an individual in its normal activities of feeding, mating and caring for young (Burt 1943; Anderson 1982; Quinn *et al.* 1992), implying the area inhabited by an individual during its complete adult life. Due to the brevity of this study, the radio-tracking data were deemed unsuitable to estimate home ranges since Quin *et al*. (1992) suggest a minimum of 36 radio-telemetry fixes is required. Rather, Utilisation areas (Bradshaw and Bradshaw 2002), which are estimates of the home range as observed for a period of less than an individual's complete adult life, were calculated using the minimum convex polygon method (Quin *et al*. 1992). GPS data depicting the radio-tagged animals' movements were overlaid on a digital aerial photograph of Scott National Park by generating new polygon shapefiles using the Esri ArcView 2.0 GIS programme. Utilisation areas were calculated after each new fix and plotted against number of fixes to produce a logarithmic curve. The logarithmic curve was extrapolated to x = 1000 to give an estimation of their home range, given a full year of radio-tracking (taking three location fixes per day), which is equivalent to the average lifespan of a *T. rostratus* (Wooller *et al.* 1981). The final utilisation value was divided by the projected value to give an estimate of the ability to predict home ranges from utilisation areas.

RESULTS

Analysis of long-term trapping data

Trapping data were available from 34 field trips over the period 1987-2006, spread seasonally as follows: summer (Dec-Feb) 15, autumn (Mar-May) 2, winter (Jun-Aug) 9 and spring (Sep-Oct) 8. Although trapping sessions were heavily biased towards the summer period, ANOVA of the mean number of trap nights per season revealed no statistically significant differences in trapping effort, with $F_{3,31}$= 0.449 and P = 0.72. Analysis by ANOVA of capture rates by site also revealed no significant differences between sites prior to the first fire in November 1993 with $F_{3,29}$ = 0.6099 and P = 0.6140. Post-fire analysis, however, revealed significant differences between capture rates in the sites with $F_{4,85}$ = 3.716 and P = 0.0078. This is primarily due to the influence of an additional trapping site 7 that was established in 1995 in the unburnt southern section of the study area and which contained extensive mature thickets of *Banksia meisnerii, Anarthria scabra* and *Adenanthos meisneri* growing to 1.5 m in height.

Seasonal Variation

Seasonal variation in capture rates were analysed separately in the unburnt sites in the southern section of study area (1, 5 and 7) from 1987 to 2006 (from 1995 in the case of Site 7), whilst those in the burnt area (4 and 6) were analysed prior to and then from 1994, immediately following the first fire in November 1993. Capture rates were substantially higher in the unburnt areas in winter, averaging 10.05 ± 1.23%, compared with non-significant values ranging from 1.5 to 2.4% in the other three seasons ($F_{3,64}$ = 22.136 and P < 0.0001). This difference was apparent and significant in all of the three unburnt sites (1, 5 and 7) whether analysed separately or pooled. This "winter effect" was not evident in the sites in the burnt area (4 and 6) and there was no seasonal effect evident in capture rates with $F_{3,39}$ = 0.9963 and P = 0.4047 on pooled data.

Impact of Fire

Trapping rates in the burnt area were, on average, lower than those recorded in the unburnt sites. Mean capture rates between burnt and unburnt sites were significantly different (burnt = 3.49 ± 0.34%, unburnt = 5.19 ± 0.63%, Mann-Whitney $U_{37,37}$ = 887, p = 0.028) and winter-capture rates were much higher in unburnt sites (burnt = 3.56 ± 0.87%, unburnt = 10.05 ± 1.23%, Mann-Whitney $U_{12, 20}$ = 214, p = $9.23\text{x}10^{-9}$). The seasonal variation in capture rates in burnt and unburnt sites is shown in Fig. 4 and Two-way ANOVA shows a strong treatment (burning) effect in winter with $F_{3,103}$ = 4.554 and P = 0.0049. The slightly higher capture rates recorded in the burnt areas in summer (3.14 ± 0.84% *versus* 1.58 ± 0.45%) was not statistically significant with t_{42} = 1.773 and P = 0.0834 and Mann-Whitney $U_{17,16}$ = 185, P = 0.081.

Impact of Rainfall

Regression analysis of capture rates *versus* prior rainfall revealed significant correlations with both annual and winter rainfall, two years prior to the trapping period. The correlations were only significant, however, in the case of winter capture rates, and then only in burnt rather than unburnt sites. Results of the regression analysis for all sites are summarised in Table 3 and the difference between the correlations between capture rates and

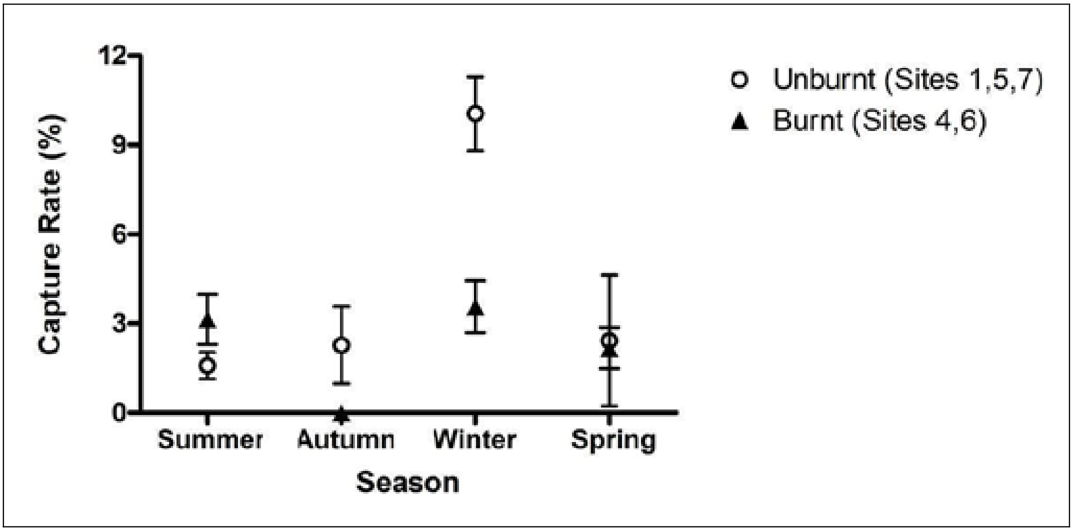

Fig. 4. Seasonal variation in capture rates, expressed as captures per 100 trap nights, of *Tarsipes rostratus* in burnt and unburnt sites of the Scott National Park study area over the period 1987-2005. Winter capture rates are significantly higher in the unburnt sites with $F_{3,103} = 4.554$ and $P = 0.0049$.

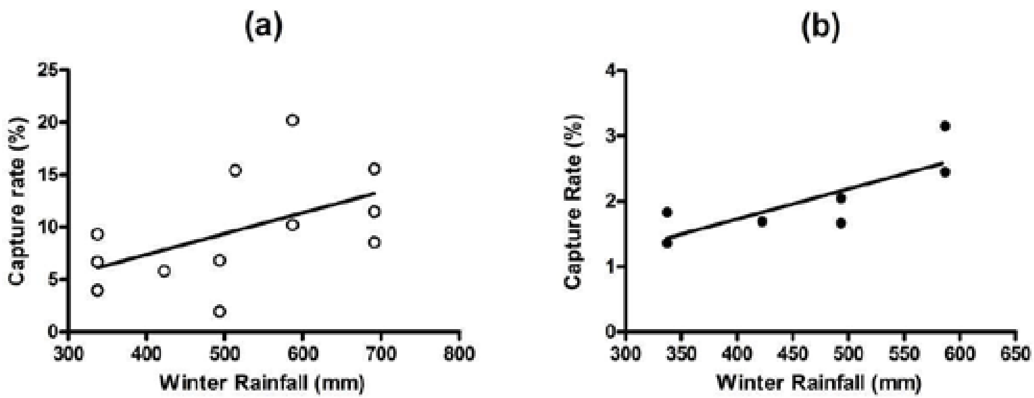

Fig. 5. Winter capture rate (expressed as captures *per* 100 trap nights) of *Tarsipes rostratus* in response to (a) winter rainfall from two years prior in unburnt sites ($r^2 = 0.258$, $P = 0.163$); (b) winter rainfall from two years prior in burnt sites ($r^2 = 0.647$, $P = 0.029$). The data are for the period 1998-2006.

prior winter rainfall in burnt and unburnt sites is shown in Fig. 5. Capture rates were not, however, correlated with rainfall from the previous year ($r^2 = 0.1155$, $F_{1,18} = 2.351$, $P = 0.1426$).

Short-term variation in trapping success

Not only are capture rates higher in winter than in other seasons of the year, but the impact of changes in weather conditions are reflected in daily capture rates. This is seen in Fig. 6 where the number of animals captured per day in two separate field trips (September 2005 and October 2006) is plotted against the recorded barometric pressure. Significant negative correlations are recorded in both cases, showing that capture rates increase substantially with the passage of a rain-bearing low pressure system.

Blossom Counts

Capture rates over the 20-year period displayed a significant positive correlation with mean inflorescence counts from *B. ilicifolia* trees as shown in Fig. 7. The equation of the regression is $y = 0.223x + 0.6442$ where y = Mean Catch Rate/Unit Effort (%) and x = mean number of inflorescences per tree for all sites with r^2 =

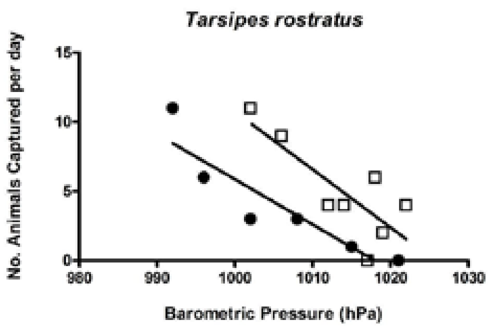

Fig.6. Correlation between daily capture rates and change in barometric pressure recorded during two separate field trips in Scott National Park: closed circles (•) September 2005, open squares (□) October 2006. The equations for the two regressions are: September 2005, $y = -0.3267x + 332.6$, $r^2 = 0.8290$, $P = 0.0107$ and October 2006, $y = -0.4178X + 428.6$, $r^2 = 0.6314$, $P = 0.0185$ where y = number of possums captured each day and x = barometric pressure in hPa. Pearson r for September 2005 = -0.9105, $P = 0.0117$ and for October 2006 = -0.7946, $P = 0.0185$.

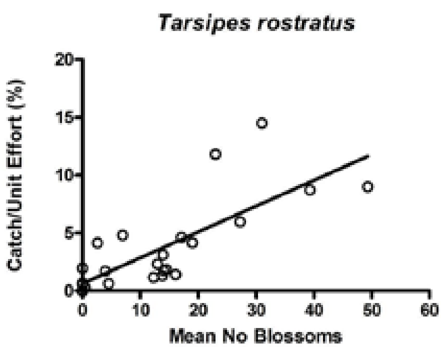

Fig. 7. Correlation between mean capture rates, expressed as captures *per* 100 trap nights, over the period 1988-2004 and mean number of fresh inflorescences counted on *Banksia ilicifolia* trees. The equation of the regression is $y = 0.223x + 0.6442$ where y = Catch/Unit Effort (%) and x = mean number of blossoms for all sites with $r^2 = 0.5737$, $P < 0.0001$ and Pearson r = 0.7574 and $P< 0.0001$ for the correlation.

0.574, $P < 0.0001$, $F_{1,21} = 28.26$ and Pearson r = 0.7574 for the correlation. Long term changes in winter and spring blossom counts for all sites are also plotted with changes in *T. rostratus* density in Fig. 8.

Long-Term Trends

Recapture rates varied between trapping sessions but were rarely high enough to attempt to estimate population sizes and data on density are used as a guide to long-term trends. Density figures from all sites and all years (Fig. 8) show large increases in the winter during the years 1996-1999 and 2002-2003. These increases closely mirror blossom counts for *B. ilicifolia*. Analysis of these pooled density data by site reveals that the first peak is due to high winter densities in Site 1 and the second in Site 7, both in the unburnt section of the study area. Long-term mean densities in burnt and unburnt sites were not significantly different (unburnt = 5.922 ± 1.274, burnt = 5.047 ± 0.718, $t_{34} = 0.598$ and $P = 0.554$) but winter densities in unburnt Site 1 have declined from the peak of 16.7 individuals.ha^{-1} in 1996 to approximately 5 from 2000 to 2006. A similar fall was evident in Site 7, the other unburnt site, with densities declining from 66.7 individuals.ha^{-1} in 2002 to 5.6 in 2005. These marked declines have been associated with a steady death from dieback of *B. ilicifolia* trees in these unburnt areas. Winter densities fell in the burnt areas following the 1993 and 1999 fires and counts of inflorescences on *B. ilicifolia* were noticeably depressed in both burnt Sites 4 and 6 for two years.

Short-term dietary and movement study - February 2004

During the February 2004 study, 22 *T. rostratus* (16 males, 6 females) were caught during 4047 trap nights, which represents an extremely low trap success (0.58 captures per 100 trap nights), even for summer survey periods. Pollen from *Banksia occidentalis*, *B. ilicifolia*, *Adenanthos meisneri*, *Beaufortia sparsa* and *Corymbia calophylla* were recorded both on the snout and in faeces. *A. meisneri* and *B. sparsa* were most abundant, being recorded on 15 of 16 and 13 of 16 individuals respectively (Table 4). These two species showed a considerable disparity in nectar sugar concentration (*A. meisneri* = 16 %; *B. sparsa* = 2 %). Despite limited availability, *B. occidentalis*, *B. ilicifolia* and *C. calophylla* were all recorded from multiple individuals. A single inflorescence of *B. ilicifolia* was observed during the study, yet pollen from this species was identified on three possums. Extensive searching for *B. occidentalis* indicated that it is limited to the eastern end of the study area, yet animals from four sites separated by approximately 4 km had *B. occidentalis* pollen on their snouts. Due to the obvious nature of the inflorescences of *B. occidentalis*, it is unlikely any were missed during the search. Despite being available, pollen grains from 16 other species of plants that were flowering at the time were not recorded from *T. rostratus*

Movements

Straight-line distances from one evening's location to the next morning's position were highly variable, ranging from 0 m to 368.3 m (mean = 69.8 ± 1.4 m, n = 49). Utilisation areas for males averaged 1.04 ± 0.20 ha (Table 5), which was not significantly greater than that reported by Bradshaw and Bradshaw (2002) across all seasons ($t_7 = 0.47$, $p > 0.05$). Movement patterns for

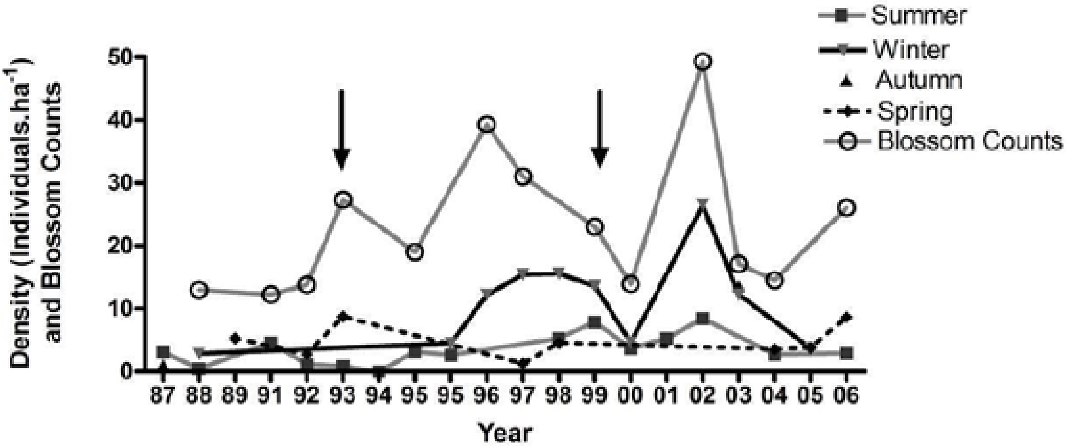

Fig.8. Long-term seasonal variation in average density measured as individuals·ha^{-1} in Scott National Park over the period 1987-2006. Mean Blossom Counts as inflorescences·tree^{-1} of *Banksia ilicifolia* are also shown for the same period. The two arrows mark the two fires that affected the northern section of the study area in November 1993 and April 1996.

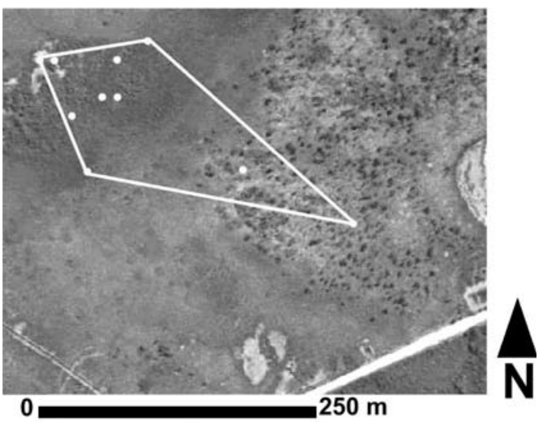

Fig.9. Movements and Utilisation Area (UA) of individual 6male (body weight = 9.2 g) over a nine-day period (17 Feb 2004 – 24 Feb 2004, 12 fixes total) in Scott National Park. UA = 2.51 ha, calculated using the minimum convex polygon method (Quinn *et al.*, 1992). Dots represent positions in which the animal was relocated. Mottled area to the east is *Banksia ilicifolia* woodland; uniform area to the west is *Beaufortia sparsa* heathland.

animal number 6male (mass 9.2 g) are shown in Fig. 9, illustrating extensive nightly movements of up to ~370 m between the *B. ilicifolia* grove, where it was first trapped, and the *Beaufortia sparsa* thicket that it used as a daytime refuge. Applying a logarithmic regression, extrapolations (x = 1000) were calculated to estimate maximum utilisation areas (an approximation of home range) for each animal. Two animals were excluded on the basis of too few data points (less than four fixes). For the remaining animals, the r^2 values were all in excess of 0.80. The utilisation areas calculated gave an estimate from 25 - 30 % of the predicted maximum utilisation area (Table 5).

DISCUSSION

Long-term Records

Long-term trapping data demonstrated pronounced seasonal variation in the apparent abundance of *T. rostratus* with capture rates being considerably higher in winter, but only in unburnt sites. The expectation of lowest abundance in summer, when food supplies would appear to be most limited, was not met. Capture rates were similar in all seasons except winter. To some extent our study suffers from limitations imposed by an opportunistic trapping protocol but we believe that the data are robust enough to reveal overall trends in the population. In addition to seasonal fluctuation in abundance, variations in rainfall are correlated with annual fluctuations in abundance of *T. rostratus* but with a marked phase difference. Capture rates in winter were significantly correlated with the amount of rain falling in the two years immediately prior to trapping, but only in the burnt areas. This suggests a delayed effect of the rain on the regenerating vegetation in the burnt areas. A similar, although slightly different result, was obtained by Wooller *et al.* (1998) working in Fitzgerald River National Park. Capture rates there were also positively correlated with rainfall, but with that of the year immediately preceding capture, rather than two years prior as we found. Wooller *et al.* (1998) attributed this correlation to the lag-time in water soaking through to the groundwater before being used by the major

Regression	Equation	Coefficient of Determination R^2	F	P	Pearson r
Annual Rainfall Unburnt	Y = 0.02048x – 10.49	0.1322	1.52	0.25	0.36
Annual Rainfall Burnt	Y = 0.004573x – 2.114	0.5525	6.17	**0.05**	**0.74**
Winter Rainfall Unburnt	Y = 0.01997x – 0.6261	0.2681	3.66	0.08	0.52
Winter Rainfall Burnt	Y = 0.004623x – 0.1226	0.6472	9.17	**0.02**	**0.80**

Table 3. Regression analysis of winter capture rates *versus* annual and winter rainfall, two years prior to capture, in burnt and unburnt areas for *Tarsipes rostratus* in Scott National Park 1998-2006.

	Habitat	BO	BI	AM	BS	CC
Number of animals with pollen (n=7)	Burnt *Banksia* woodland	2	0	7	5	1
% with pollen grain		28.57	0	100	71.43	14.29
Number of animals with pollen (n=4)	Burnt heath	3	2	3	4	0
% with pollen grain		75	50	75	100	0
Number of animals with pollen (n=5)	Unburnt *Banksia* woodland	2	1	5	4	1
% with pollen grain		40	20	100	80	20

Table 4. Pollen grains collected from *Tarsipes rostratus* in Scott National Park. Habitat refers to the habitat in which the animal was captured. Pollen grain types: BO = *Banksia occidentalis*, BI = *Banksia ilicifolia*, AM = *Adenanthos meisneri*, BS = *Beaufortia sparsa*, CC = *Corymbia calophylla*.

Animal	Projected Home Range (x = 1000)	Utilisation Area (ha)	N	% Home Range Observed	Animal Mass (g)
4M	2.10	0.53	12	0.25	6.9
5M	0.92	0.28	14	0.30	6.5
6M	9.08	2.46	10	0.27	9.2
11M	-	0.46	3	-	9.5
12M	3.24	0.91	9	0.28	7.1
16M	-	0.84	5	-	10.5

Table 5. Utilisation areas and home range approximations for *Tarsipes rostratus* at Scott National Park. Home ranges for animals 11M and 16M were not calculated due to insufficient data points. N = number of fixes.

food plants of *T. rostratus* (Dodd *et al*. 1984). This relationship was not evident, however, during summer trapping in the present study, nor in the long unburnt sites within the study area, and the two-year effect we have revealed appears to be specific to the vegetation regenerating after fire.

Another significant short term effect of weather to emerge from this study is the increase in capture rates seen with the passage of a cold front. Data were only available from two trapping periods in 2005 and 2006 but, in both cases, the passage of an intense low-pressure rain-bearing depression, with barometric pressures below 1000 hPa, was associated with elevated capture rates of *T. rostratus* with strong negative correlations. Published information on the effect of barometric pressure on animal behaviour is meagre, although there are anecdotal records of animals being influenced by such changes (Stokes, Slade and Blair 2001). Chapman and Trethewey (1965) reported a positive correlation between barometric pressure and captures per trap night in eastern cottontail rabbits, *Sylvilagus floridanus,* in western Oregon and Hoff and Hillyard (1993) found that changes in barometric pressure affected rates of cutaneous water uptake in dehydrated desert toads, *Scaphiopus couchi*. Bats have also been reported to use barometric pressure as a cue when tracking insects (Paige 1965). In the case of *T. rostratus*, falling barometric pressure obviously presages rain and nectar production by plants is known to be stimulated by high humidity and rain (K. Dixon pers. com.) The increased behavioural activity associated with foraging for this nectar clearly leads to a greater probability of animals falling into a pitfall trap.

In addition to rainfall, fire history has a significant effect on the abundance of *T. rostratus* as reflected in capture rates and density values. The significant difference in winter capture rates and densities between burnt and unburnt *Banksia* woodland, suggests a greater resident population in the unburnt habitat. This is likely to result from the response of their food plants to fire. *B. meisneri*, one of few winter-flowering potential food species in the area, is killed by fire and regenerates from seed (George 1987) resulting in a poorer food supply in burnt areas during the winter months post-fire, while losses of *B. occidentalis* and *Beaufortia sparsa* reduce the food availability during late summer and early autumn. The food plants of *T. rostratus* vary in their response to fire (dying off and regenerating from seed or regenerating from epicormic shoots or lignotubers), thus the time since a fire will have a differential effect on their flowering (Bell *et al*. 1984). Capture rates were noticeably similar in burnt and unburnt areas in summer and this appears to be due to the extensive flowering of *A. meisneri* several years after fire. Radio-tracking data show that individuals were routinely moving from the unburnt side of the road to the burnt heath to feed in summer, but were not resident in the burnt habitat. Therefore, the abundance of *T. rostratus* post fire is an interaction between the response to fire of their food species and the phenology of flowering as they regenerate. *T. rostratus* populations would be most susceptible to a fire régime of frequent, extensive fires when their food species are killed by fire. Estimates of the time needed for populations of *T. rostratus* to recover after fire vary from 15-25 years (G. Friend, unpub., Everaardt 2003).

Feeding behaviour and movements of possums

Given that flowering specimens of the species and Families of plants believed to be *T. rostratus*' exclusive food source were of low abundance during the late-summer study period, an obvious expectation is that the animals might exploit novel food plants beyond these three Families. Our identification of pollen grains provides no evidence that this occurred. *T. rostratus* fed only on Myrtaceae, Proteaceae and Epacridaceae species. Within these groups they consumed species that have a floral morphology that is also amenable to bird pollination (Brown *et al.* 1997).

Our radio-tracking data, coupled with the pollen records, show that *T. rostratus* move extensively between neighbouring habitats and can travel substantial distances to locate restricted food sources. Based on identification of pollen grains, it is likely that several individuals travelled in excess of a kilometre to reach small stands of *B. occidentalis* in late summer. Less extreme is the movement of almost all individuals between *Banksia* woodland and heath to obtain *A. meisneri* and *Beaufortia sparsa*. These movements may be driven by selective foraging for pollen and nectar from different plants, as first suggested by Bradshaw and Bradshaw (1999) based on the absence of any correlation between nectar and pollen intakes for individual animals. The sugar concentration of *B. sparsa* flowers, for example, was eight times lower than that of *A. meisneri*. Therefore, it would be more profitable for *T.rostratus* to use the latter species as a nectar source. On the other hand, the stamens of the *Beaufortia* flower are a prominent part of the flower, which indicates they may have more pollen available.

Utilisation Areas of males tracked in this study, averaging 1.04 ± 0.49 ha, were not significantly larger than the figure of 0.79 ± 0.24 ha previously reported for male *T. rostratus* across all seasons in Scott National Park (Bradshaw and Bradshaw 2002). The utilization areas observed in this study indicate that the summer reduction in food resources is not accompanied by an obvious increase in the size of the animals' foraging area. The home ranges estimated from our data set suggest, however, that *T. rostratus* may exploit areas much greater than previously believed and further tracking of *T. rostratus* in different habitats and at different times of the year would be most desirable. The very much larger areas over which *T. rostratus* was found to move in Scott National Park is in contrast to the much smaller figures for home range given by Garavanta *et al.,* (2000) for the same species in Fitzgerald River National Park. Their data, however, are based on trapping data alone and Bradshaw and Bradshaw (2002) show that there is a large disparity between such trapping data and that obtained by radio-tracking of individuals. Habitat differences may also play a part, however, and the extensive more open burnt areas in Scott National Park may be a factor influencing daily movements. Plant distribution in the Fitzgerald River National Park is also more uniform that in Scott National Park, where food sources tend to be clumped, and this may also be a factor influencing more extensive movements in the latter habitat.

Physiological adaptations may also play a role for *T. rostratus* with restricted food availability, namely using metabolic reduction in the form of torpor to reduce their energy requirements. During the study period there were two instances of animals with body temperatures recorded close to ambient (Phillips *et al*. 2004) which is suggestive of torpor. Previous workers have found torpid *T. rostratus* in pit traps and recorded torpor in the laboratory when animals are exposed to a combination of low temperatures and restricted food (Collins *et al*. 1987; Withers *et al*. 1990). Nagy *et al.* (1995) also suggested that a marked individual variation in the field metabolic rate (FMR) of *T. rostratus* measured in Fitzgerald River National Park may result from the use of torpor.

CONCLUSIONS

This study provides strong evidence that season, yearly variation in rainfall, and fire history all play an important part in determining the apparent abundance of *T. rostratus* populations. Given the evident link between rainfall and food availability for an obligate nectarivore, it can be predicted with a high degree of certainty that the long term reduction in rainfall currently being experienced by south-western Australia (Smith *et al.* 2000) will have a negative impact on *T. rostratus* populations. This could also be exacerbated by the impact on the vegetation of local removal of water from underground aquifers (Groom *et al*., 2000a). Thus, proposals such as that of the Water Corporation of Western Australia to abstract 45 GL of water per year from the South West Yarragadee Aquifer (Water Corporation of WA, 2005) need to be assessed in terms of impacts to the ecology and conservation of species that are not intuitively linked to such resources.

The impact of fire will depend on the plant species present, their response to fire, and elements of the fire régime such as extent, frequency, and time of year. The most harmful régime would be extensive fires at frequent intervals that can result in local extinctions of their preferred food species (Meney *et al*. 1994). Furthermore, the inflexibility of their diet will leave them susceptible to the effects of *P. cinnamomi*, to which approximately 40% of the flora is susceptible, including numerous taxa in the Epacridaceae, Myrtaceae and Proteaceae (Shearer *et al*. 2004). Our results suggest that the persistence of the *T. rostratus* population in Scott National Park may be highly dependent on the management of the ecosystem. Effective management of the *T. rostratus* will require careful consideration of the interaction of fire régime with the effects of long-term rainfall patterns and potential external impacts such as water extraction and the spread of *Pa cinnamomi*.

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