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Presence of the Dingo (*Canis lupus dingo*) on Risk Sensitive Foraging of Small Mammals in Forest Ecosystems

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Presence of the dingo (*Canis lupus dingo*) on risk sensitive foraging of small mammals in forest ecosystems

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Abstract

Trophic regulation of mesopredators through top order predators can have profound effects on ecosystem community and diversity. In the absence of top predators, invasive mesopredators exert strong selective pressures on native prey and can alter prey foraging behavior. When foraging in the presence of predators, prey must weigh predation risk against food gain. To examine the indirect impacts of dingo baiting on risk sensitive foraging in forests, we measured differences in giving up densities (GUDs) and surveyed local populations of mesopredators and mammals. We hypothesized that in baited areas, mesopredators would be more abundant and prey would perceive greater predation risk. Foraging trays of peanuts were placed in baited and nonbaited study areas for four nights and the remaining peanuts measured as the GUD. A higher density of mesopredators and a lower density of small mammals was observed in baited versus nonbaited study sites. Consistent with foraging theory, rodents perceived significantly greater predation risk in baited areas than nonbaited areas. However, abundance of medium and large mammals was not affected by baiting regime. Ecosystem conservation management has strongly focused on baiting of invasive predators in protected reserves. Our study suggests removal of a top predator positively affects mesopredator abundance and negatively affects small mammals in forests. For sustainable forest management, reconsideration of baiting regimes may be necessary to optimize ecosystem diversity and structure.

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1. Introduction

The presence of top predators in animal communities has important consequences for ecosystem structure and function. Predators can exert a top-down effect on herbivores and smaller predators through traditional trophic cascade theory. Suppression of herbivores can positively impact plant species and diversity while intraguild predation or competition can promote small mammal and bird diversity (Ripple and Beschta 2002). The keystone role of top-order predators amplifies their impact on prey populations and habitat diversity.

Predation can include both direct effects on populations through consumption and selective pressures on prey behavior. Top order cascade effects can alter behavior to reduce prey activity or shift activity to safer habitats. Under optimal foraging theory, prey must weigh feeding rate against predation risk on whether to utilize a feeding patch (MacArthur and Pianka 1966). When food density is high and predation risk low, prey benefit by continuing to forage in the harvest patch. However, when food density is low and predation risk is high, prey species must increase vigilance and predation risk may outweigh any benefit from foraging. Testing foraging behavior of prey requires measuring habitat use, preferences, and the acceptance or rejection of patches (Brown et al. 1988).

While risk foraging preferences can be directly influenced by predation risk (Brown et al. 1992; Hughes and Ward 1993; Jacob and Brown 2000), the effect of top order trophic cascades on foraging behavior has only been recently studied (Letnic and Dworjanyn 2011; Strauß et al. 2008). Under the mesopredator release hypothesis, removal of top predators has a direct positive effect on mesopredators and an indirect negative effect on prey diversity (Crooks and Soule 1999). Top predator interaction with mesopredators can include opportunistic intraguild predation on mesopredators or mesopredator avoidance of areas frequented by top predators due to perceived predation risk. Consequently, following removal of top predators, changes in mesopredator behavior can include strong predation pressure on

prey and lead to local extinction (Johnson et al. 2007). Former studies on the mesopredator release hypothesis have focused on the ecological role of wolves in North America (Hebblewhite et al. 2005). In wolf absent areas, elk populations were greater and herbivory reduced vegetation complexity and structure, which subsequently affected songbird diversity.

In Australia, the dingo has been intensively studied as a keystone species because of its predatory and competitive effects on invasive mesopredators (Glen et al. 2007). One of Australia's most ecologically influential predators, the dingo arrived approximately 3500-4000 years ago with Asian seafarers (Corbett 1995). While originally brought as human companions, dingoes frequently escaped and quickly established wild populations. Soon after the introduction of the dingo, both native marsupial predators, the thylacine and Tasmanian devil, went extinct on the mainland possibly due to outcompetition from the dingo. With the conversion of land to pastoral use, dingoes have been driven from much of southeastern Australia (Glen and Short 2000). Hundreds of additional introduced species have contributed to the extinction or demise of native flora and fauna. Invasive mesopredators such as the European red fox (*V. vulpes*) and the house cat (*F. catus*) are widespread across the continent and have significant impacts on biodiversity of small and medium-sized mammals.

Inverse relationships between dingo and fox activity have been found in southeastern Australia (Newsome et al. 1997). The survival of native marsupials has been closely correlated with high dingo density supporting top predator maintenance of prey biodiversity (Johnson et al. 2007). A parallel study on the impacts of dingoes in arid Australia, Letnic et al. (2009) found reduced mesopredator abundance and reduced herbivore activity in the presence of dingoes than in the absence of dingoes when compared across the NSW dingo fence. Within the last century, pastoral interests have led to heavy dingo baiting and subsequent removal of a top predator.

Forest ecosystems in southeastern Australia hold a rich diversity of flora and fauna. Wild dingo populations persist in areas along the coast while in other areas, ongoing baiting suppresses populations. Previous regional studies have investigated dingoes and competitive exclusion foxes in New South Wales forests although human presence was attributed as a dominant factor in determining the distribution of foxes (Catling and Burt 1995). Evidence in support of a negative relationship between fox and dingo abundance in eastern forests suggests abundance of dingoes sets an upper limit on the abundance foxes (Johnson and VanDerWal 2009). Extensive scat analysis of dingo and fox diet in the Sydney area has shown foxes consume a greater range of prey than dingoes and a greater proportion of their diet is comprised of small and medium sized mammals, birds, reptiles, and insects (DECC 2007). Intraguild predation and dietary competition have also been suggested to suppress foxes (Cupples et al. 2011).

Although risk sensitive foraging in relation to dingo presence has been studied in arid Australia (Letnic and Dworjanyn 2011), little is known about the impacts of dingo presence on rodent foraging in forest ecosystems. Here, we study the indirect effects of dingo presence on mesopredator abundance and rodent foraging behavior in eastern New South Wales forests. We measured predator and prey abundance and perceived predation risk in baited and nonbaited areas. First, since dingoes populations will be suppressed in baited areas, a higher density of mesopredators is expected based on the mesopredator release hypothesis. Secondly, greater density of mesopredators will exert greater predation pressure on small mammals and influence foraging behavior of prey. We predict increased perceived predation risk in baited areas in accordance with optimal foraging theory.

2. Methods

2.1 Study sites

Due to pressures from domestic livestock and urban settings, the presence of the dingo in NSW is now limited to the coastal ranges and the northwest corner of the state. Two adjacent study areas were selected in the mid-North Coast to assess prey foraging behavior in the presence and absence of dingoes: Hat Head National Park (31°2'27''S, 153°1'35''E; baiting) and Limeburners Creek Natural Reserve/Goolawah National Park (31°18'8''S, 152°53'49''E; no baiting) are forested habitats on the coast that form part of system of eight coastal protected areas from the Manning River to Coffs Harbor. The parks were selected based on similar vegetation, habitat, landforms, and close proximity to minimize climactic factors (10 km apart). Hathead National Park contains 7,200 hectares of coastal land with vegetation communities of wet and dry heathland, littoral rainforest, eucalypt forest, and woodland (Hat Head NP Plan of Management 1998). Mean annual rainfall in the park is 1484.0 mm (1939-2011; Australian Bureau of Meterology). Intensive 1080 baiting in the past and currently in Hat Head has retarded dingo presence. Limeburners Creek NR was the first nature reserve declared on the north coast of New South Wales and covers an area of 9,123 hectares of wet and dry heathland, littoral rainforest, eucalypt forest and woodland, as well as fresh water and estuarine wetlands (Limeburners Creek NR Plan of Management 1998). Mean annual rainfall at Port Macquarie, 12.6 km away, is 1534.5 mm (1840-2010; Australian Bureau of Meterology). As part of the landcare initiative the Maria River Project, no dingo or fox baiting has taken place in Limeburners NR or adjoining properties since 2007. The park has a stable dingo population that is closely monitored through satellite tracking (NPWS 2011). Goolawah NP is an adjacent regional park of 534 hectares created in 2010 with similar vegetation and managed by Limeburners Creek NR (no baiting regime).

Within each study area, eight sites were spatially placed along roads at least 1 km apart (Figures 1 and 2). Baited sites were placed several kilometers distant from nonbaited sites.

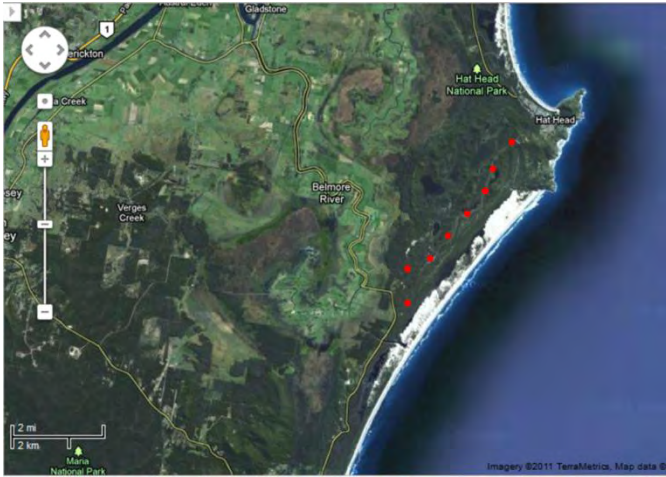


Figure 1 Map of Hat Head National Park. Dots indicate locations of each site (Source: Google Maps).

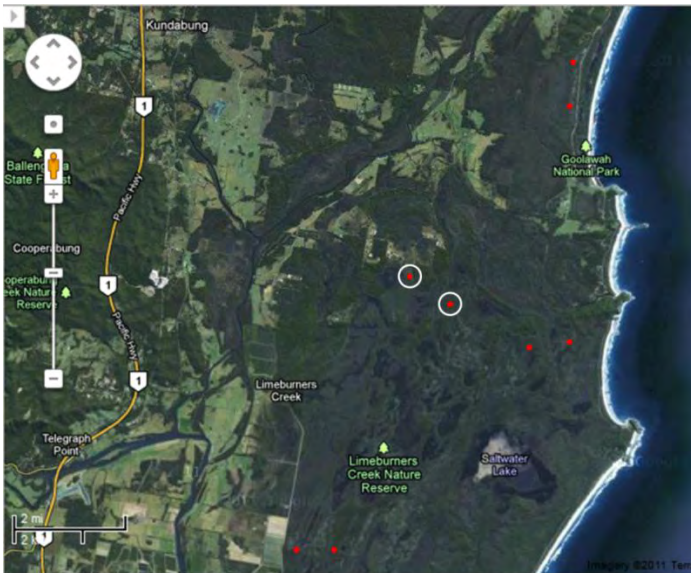


Figure 2 Map of Limeburners Creek Nature Reserve and Goolawah National Park. Dots indicate locations of each site. Two sites (circled) were located on private land adjacent to Limeburners Creek NR with the permission of the owners Goolawah Co-op (Source: Google Maps).

2.2 Experimental Design

Mammal foraging activity was assessed by measuring giving up densities (GUDs) through foraging trays in the eight sites for each study area. Under optimal patch use theory, resource depletion during foraging reduces the harvest rate and thus the benefit for the prey. Declining harvest rates thus must be weighed against metabolic cost, predation risk, and missed opportunity costs of foraging (Brown 1988). Assuming stable metabolic and opportunity costs between study areas, we predict that dingo regulation of mesopredator

abundance should mediate predation risk. Since the density of food decreases as more food is harvested, the giving up density is a measure of the prey perception of foraging costs and predation risk. This method has previously been established as a quantitative measure of perceived predation risk (Brown et al. 1997; Letnic et al. 2009).

Four artificial food patches were established at each of the sites using aluminum trays (30 x 20 x 5 cm) provisioned with 20 peanut quarters each and randomly mixed with 1 L of sifted sand to create an even distribution and declining gain. The number of seeds used was replicated from previous work in arid Australia (Letnic and Dworjanyn 2011). To attract medium-sized mammals such as long-nosed bandicoots (*Perameles nasuta*), peanuts quarters were coated with black truffle oil (Paull et al. 2010). Peanut trays were placed for four nights at each site. The first experiment was run at Hat Head for four nights (15-21 November 2011, heavy rains forced us to suspend experiments for two nights) and the second experiment was run at Limeburners Creek/Goolawah the following week (22-25 November 2011) for four nights. Both experiments were conducted under a waning moon. We also trialed mealworms as a bait attractant for medium-sized mammals in two food patches at each site for two nights of each experiment (Searle et al. 2008). Ten mealworms (larval stage of *Tenebrio molitor*) were placed in aluminum trays (22 x 16 x 5 cm) and mixed into 1 L of sand (Kovacs et al. 2011).

The area around each peanut or mealworm tray was swept so footprints could be observed. Each morning foraging trays were checked for signs of disturbance or tracks and the number of peanuts or mealworms remaining was counted as the giving up density (GUD). Foragers were identified by inspecting the tracks present on the sand or the chew marks on the nuts. Confirmation of the species that visited the trays was conducted by mounting an infrared game camera with movement sensors on one tray at each site (ScoutGuard). Cameras were set to 1 min video with 1 second interval from 6:00 PM to 7:10 AM the following day.

Missing peanuts and dead or missing mealworms were replaced every morning and surrounding sand swept.

Mean giving up densities were calculated as the mean number of peanut or mealworm pieces remaining per night per tray and analyzed using a generalized linear model with study area (baited or nonbaited) as the variable factor through SPSS Statistics 18.

2.3 Small mammal trapping

To estimate abundance of small mammals in each study area, trapping grids were laid out at each of the eight sites. A total of 20 traps within each site were situated 10 m apart in a 50 m x 50 m square. All trapping grids were positioned approximately north-south at least 10 m away from the road. Folding aluminum Elliott traps (33 x 10 x 10 cm) were baited for four consecutive nights with a mixture of rolled oats, peanut butter, honey, and truffle oil. Traps were checked from first light every morning. Captures were identified, weighed, and sexed. After first capture, individuals were marked on the tail with permanent marker to identify recaptures.

2.4 Predator activity

To assess presence of predators in the area, 16 sand plots at Hat Head and 16 sand plots at Limeburners/Goolawah were placed at 500 m intervals on the road. An area the width of the road x 1 m was swept each afternoon and the tracks observed and identified the following morning. Sand plots were maintained for four consecutive days at each study area (baited and nonbaited).

2.5 Faunal abundance

In order to consider the predatory impacts on prey, we compared abundances of prey populations in each study area through transect runs. Macropod and faunal abundance activity was measured through afternoon surveys. Two afternoon surveys in each study area were conducted between the hours of 5-8:00 PM on transects ranging from 9 m to 11.1 km

along the road while driving at 15 km/h. Visibility along the road was 10 m to each side and an index of abundance was calculated as the mean number of animals sighted per a kilometer surveyed. We conducted three spotlighting surveys from ranging from 6.3 to 9.1 km in each study area from a four wheel drive traveling at 5 km/h. Visibility was 10 m on each side of the road. Abundance data was calculated as the mean number of animals sighted per a kilometer surveyed.

At each of the eight sites in both study areas, four scat surveys were undertaken on linear transects of 100 m length. The observer looked one meter to each side and all scats were tallied and identified. Indications of animal presence were also counted on each transect (goanna diggings, burrows, scratchings etc).

2.6 Medium-sized mammal camera trap

Activity of medium sized mammals was monitored using baited camera traps (ScoutGuard). A bait of honey, oats, peanut butter, and sardines was placed in a tube cylinder on the ground with openings. Mounted cameras were set to 15 second video recordings from 6:00 PM to 7:10 AM and collected after 4 nights for identification. Two camera traps were placed at each site.

2.7 Habitat complexity

To account for possible differences in vegetation between study areas, an assessment of habitat complexity at all eight sites in both study areas was conducted. Percentage canopy covered was measured by walking 200 m around the trapping grid in the site and recording canopy cover or sky every 5 meters. At every 10 m, the number of understory layers and maximum height within a 1 m radius was recorded. In order to measure the vegetation density, a horizontal coverboard method was used (Monamy and Fox 2000). A checkered 20 cm x 50 cm coverboard of ten 10 cm x 10 cm squares was held 5 meters away from an observer at 0–20 cm; 20–50 cm; 50–100 cm; 100–150 cm; and 150–200 cm in height. Ground

cover was also estimated by taking 50 steps within a 10 m x 10 m plot and recording green vegetation, leaf litter, log, or bare ground at each step. Four estimates of coverboard and ground cover methods were made at each of the north, east, south, and west points of the trapping grid.

3. Results

3.1 Foraging behavior

Takes by rodents from the peanut foraging trays were recorded at both the baited and nonbaited study areas (n=17, baited; n=23, nonbaited). Mean giving up densities were significantly lower in nonbaited study areas compared to baited study areas with $p < 0.05$ (Figure 3). Two medium-sized mammal takes at peanut trays were recorded but not included in calculations due to variations in giving up density for medium versus small mammals. Camera footage confirmed rodent foraging at trays overnight.

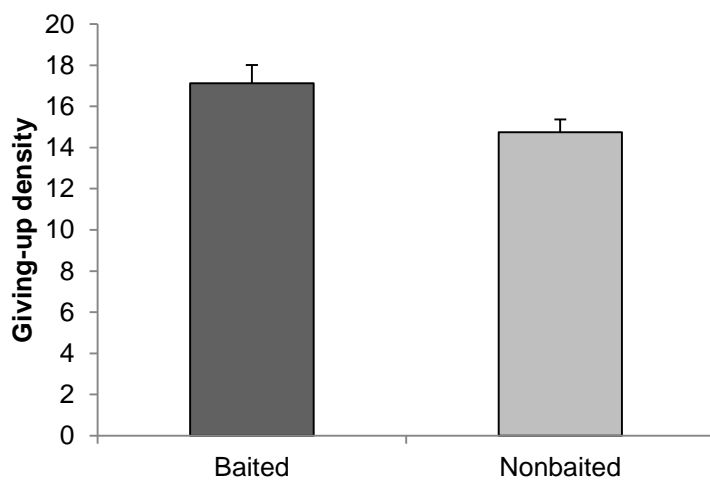


Figure 3 Mean peanut giving up densities (\pm SE) of rodents in nonbaited and baited areas. The giving up density represents the number of peanuts remaining in 1 L of sand.

Foraging from mealworm trays was not observed from any sites. Elevated precipitation during foraging nights and ant predation reduced survival of mealworms. Detection of mealworm escape or bird foraging was also difficult. However, the mean

number of mealworms remaining was calculated as 6.27 ± 0.42 SE in baited and 7.52 ± 0.40 SE in nonbaited sites.

3.2 Small mammal capture rate

We calculated an index of diversity for small mammals using the Brillouin index of diversity (Brillouin 1956) using the following equation:

$$H = \frac{\ln N! - \sum \ln n_i!}{N},$$

where H is diversity, N is the total number of individual captures, and n_i is the number of individual prey items in the i th category. The Brillouin index of diversity was used instead of the Shannon index because of the uncertainty that exists regarding the species to which an individual selected at random belongs and its validity when the number of species caught is low. The Brillouin index has been used as a comparative measure of rodent trapping grids in Australia (Read et al. 1988). The Brillouin index of diversity was 0.295 for baited and 2.285 for nonbaited sites indicating a higher diversity in nonbaited sites. Four species were captured in total (*R. fuscipes*, *R. rattus*, *R. lutreolus*, *A. stuartii*). Mean capture rates (captures per trapping night) were higher for all species in nonbaited study areas compared to baited study areas (Figure 4).

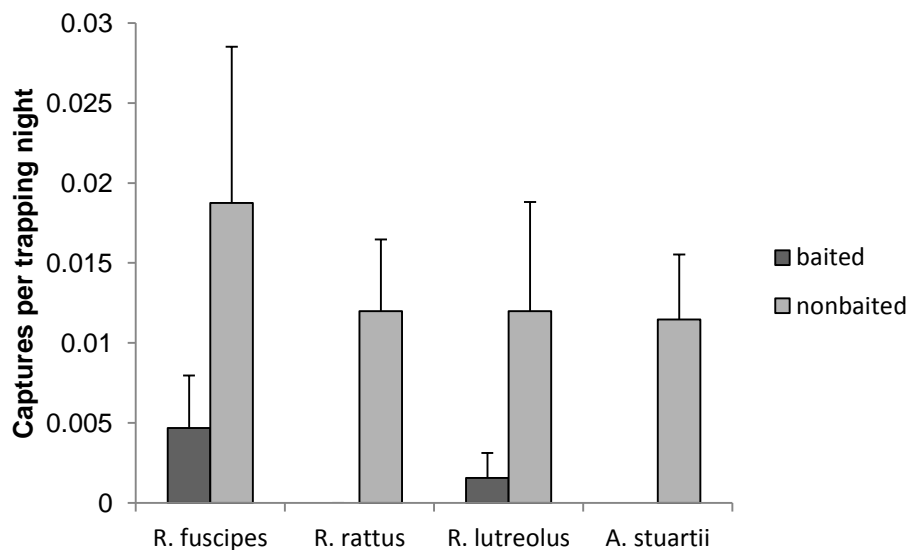


Figure 4 Mean abundance (\pm SE) of small mammals in captures per trapping night. Trapping nights were counted as the number of traps open per a night.

3.3 Predator sand plots

Dingoes were detected on plots at both the baited and nonbaited sites; however, the mesopredators cat (*F. catus*) and fox (*V. vulpes*) were detected only at the baited sites (Figure 5). Eastern grey kangaroos (*Macropus giganteus*) were more frequently detected in nonbaited sites while medium-sized marsupials had comparable densities at both study areas.

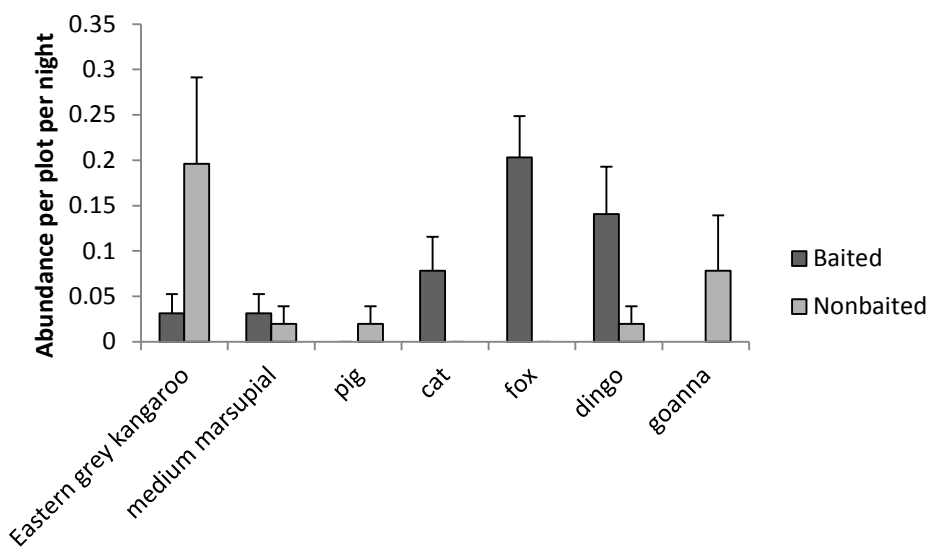


Figure 5 Mean abundance (\pm SE) on a predator sand plot per a night over four nights.

3.4 Medium-sized mammal camera traps

Fauna detected by the camera traps was calculated as the total number of individuals observed over a four night period with 16 cameras in each study area (baited or nonbaited). Two species were observed at the baited site and three species were observed at the nonbaited site. Overall, abundance of fauna recorded was higher at nonbaited than baited sites (Figure 6).

	Baited	Nonbaited
<i>Rattus rattus</i> (black rat)	1	0
<i>Rattus fuscipes</i> (bush rat)	0	5
<i>Tachyglossus aculeatus</i> (echidna)	0	2

<i>Trichosurus vulpecula</i> (common brushtail possum)	0	2
<i>Felis catus</i> (house cat)	1	0

Figure 6 Faunal abundance observed by infrared cameras. Total number of individuals recorded at 16 cameras over four nights.

3.5 Scat and trace count

Searches for scats and other traces yielded a greater diversity of animals per hectare at nonbaited than baited sites. Mean abundance was calculated by averaging the number of scats and traces per hectare. Abundance of medium-sized mammals: wallaby (*M. rufogriseus* or *W. bicolor*), bandicoot (*P. nasuta*), potoroo (*P. tridactylus*), echidna (*T. aculeatus*), sugar glider (*P. breviceps*), koala (*P. cinereus*), or wombat (*V. ursinus*) was higher at nonbaited than nonbaited sites. However, abundance of macropods was slightly higher in nonbaited areas. Dingo scats were found only in the nonbaited study area.

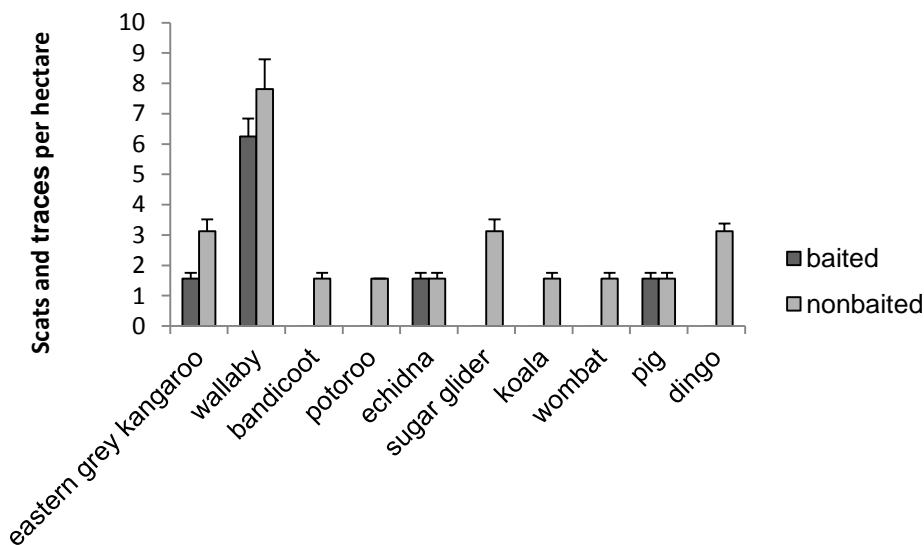


Figure 7 Mean number (\pm SE) of scats or traces per a hectare in baited and nonbaited areas.

3.6 Spotlighting

Spotlighting transects were divided by the number of kilometers surveyed to calculate average abundance/km. Eight species were observed in total (6 in baited and 4 in nonbaited).

Overall, abundance of medium-sized prey was similar in baited sites and nonbaited sites. Cats were observed in baited sites but not in the nonbaited sites.

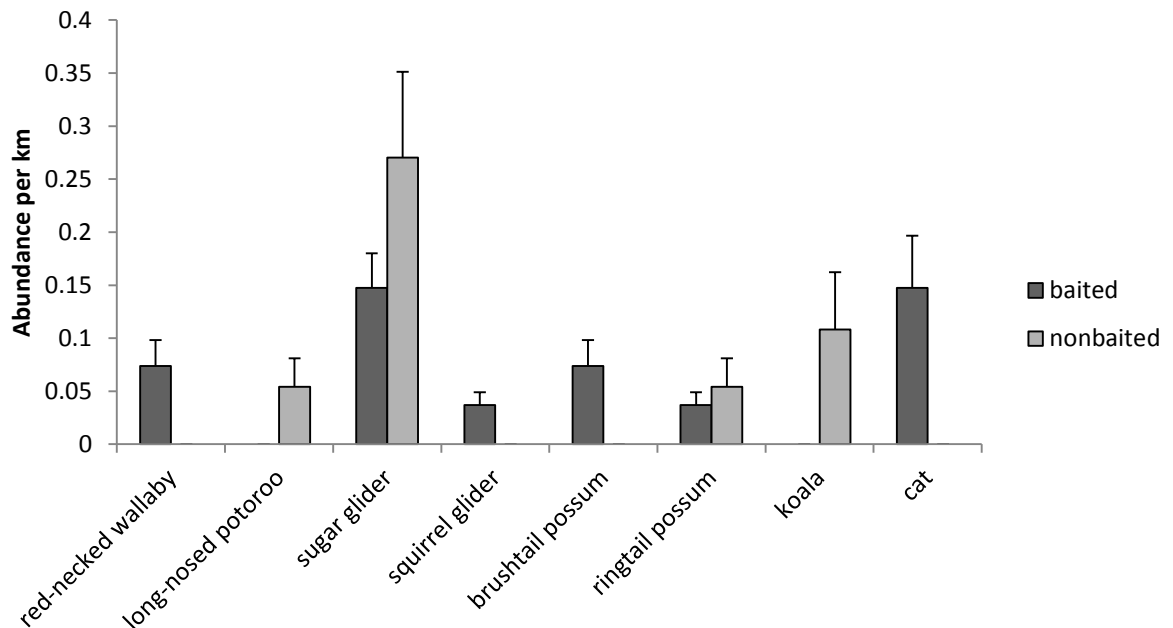


Figure 8 Mean abundance (± SE) per km surveyed by spotlight. Three spotlights were conducted in the baited area and only two spotlights were conducted in the nonbaited area due to inclement weather.

3.7 Afternoon surveys

Abundances of fauna were calculated as the number per a km surveyed. The mean density of macropods sighted was much higher in baited than nonbaited sites.

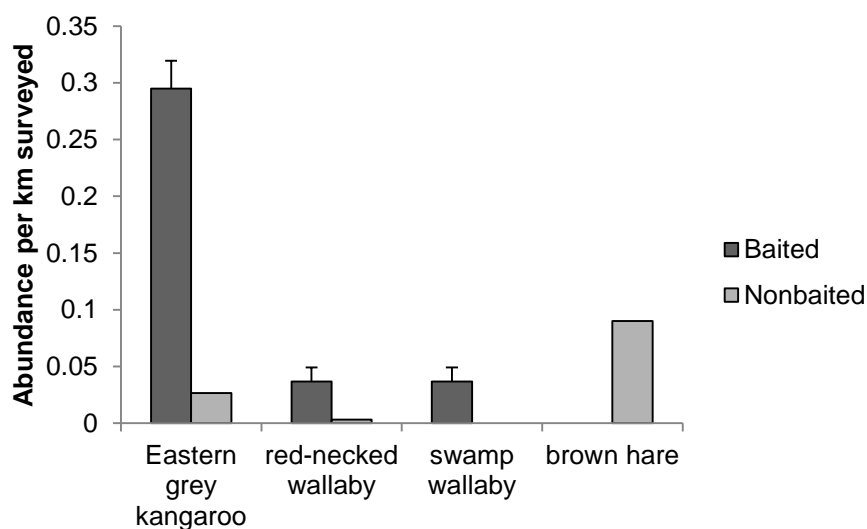


Figure 9 Mean abundance (± SE) per km surveyed in the afternoon. Only one survey was conducted at the nonbaited site due to inclement weather.

3.8 Habitat Assessment

Vegetation density calculated by the coverboard method was similar in both the baited and nonbaited study areas (Figure 9). Percent visibility at all heights except 150-200 cm was slightly higher in the baited sites. Ground cover composition was similar between both study areas and dominated by leaf litter followed by green vegetation, bare ground, and logs. However, the baited site was composed of more leaf litter $58.2 \pm 3.4\%$ than the nonbaited $48.6 \pm 3.2\%$ while the proportion of green vegetation was greater in nonbaited than baited areas (Figure 11). Canopy cover was greater in unbaited sites $68.5 \pm 6.4\%$ compared to $59 \pm 3.2\%$ in baited sites (Figure 12). However, understory complexity measured by mean understory layers and height was similar between study areas.

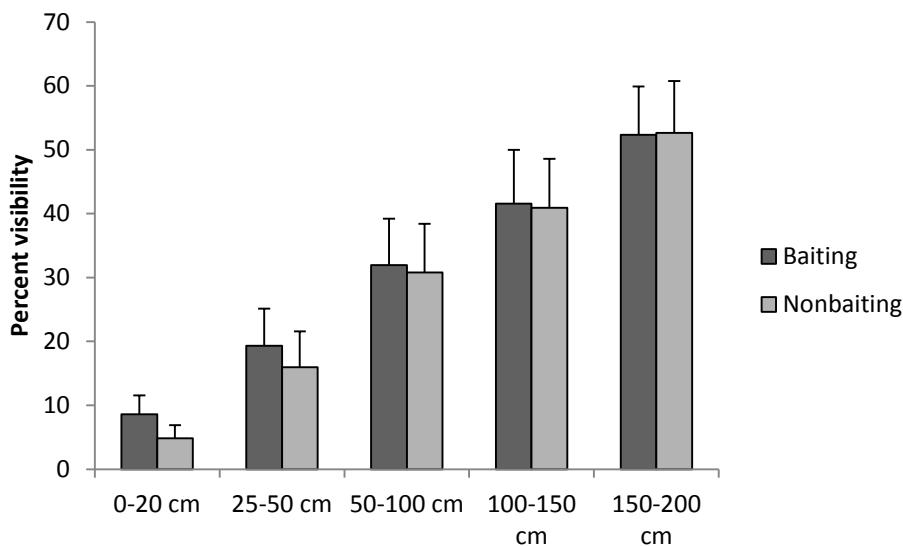


Figure 10 Mean percent visibility (\pm SE) through vegetation at varying height intervals above ground. Mean visibility at each study area was calculated by averaging the proportion of squares visible out of 10 squares on a 50 cm x 20 cm checkered coverboard.

	Baited	Nonbaited
Litter	58.2 ± 3.4	48.6 ± 3.2
Green vegetation	32.3 ± 3.6	43.6 ± 4.5
Bare ground	6.9 ± 1.9	4.2 ± 2.2
Log	2.6 ± 0.4	2.4 ± 0.9

Figure 11 Mean percent (\pm SE) ground cover composition in baited and unbaited study areas.

	Baited	Unbaited
--	--------	----------

Canopy	59 ± 3.2 %	68.5 ± 6.4 %
Open	41 ± 3.2 %	31.5 ± 6.4 %
Number of understory layers	2.57 ± 0.12	2.41 ± 0.06
Understory height	1.99 ± 0.09 m	2.11 ± 0.09 m

Figure 12 Mean percent canopy and understory complexity (\pm SE) in baited and unbaited study areas.

4. Discussion

The results from this study support the mesopredator release hypothesis with increased mesopredator activity in areas of baiting and subsequent suppression of small mammal abundance. These findings are consistent with previous studies on the negative correlation found between dingoes and mesopredators (Kennedy et al., 2011; Letnic and Dworjanyn 2011). In accord with the expectation that perceived predation risk is higher with greater predator density, giving up densities of rodents were higher in the baited study area. However, while small mammal abundance demonstrated direct impacts of predation, medium-sized mammal and macropod abundance appeared to be unaffected by dingo presence.

Lower abundance and diversity of small mammals in baited areas was most likely indicative of increased mesopredator activity. Presence of *V. vulpes* and *F. catus* on predator track plots was observed only in baited study areas. In studies where foxes have been removed, increases in small mammal populations have been observed (Kinnear et al. 2002; Dexter and Murray 2009). Furthermore, evidence of native mammal and bird predation by *F. catus* and subsequent decline of local populations has been well documented (Dickman 1996). The exclusion of mesopredators by dingoes has been supported by studies of interactions between feral cats and dingoes, which suggest that distributions of feral cat

populations in dense cover may reflect the use of cover as more protection from *C. lupus dingo* (Edwards et al. 2002).

The presence of *C. lupus dingo* tracks on sand plots in both study areas indicate that baiting regimes may not completely exclude dingoes from the area. Dingo tracking satellite data in the nonbaited sites (NPWS unpublished) indicate that while all dingoes tracked spent the majority of time within the Limeburners Creek Nature Reserve (nonbaited park), dingoes travel between the two parks. Therefore, our model may not reflect strictly dingo present and absent ecosystems, but rather greater dingo activity in the unbaited study area. However, dingoes that do travel through Hat Head National Park (baited) frequently die from bait consumption. Accordingly, Limeburners Creek NR sustains a stable dingo population whereas Hat Head NP does not. While dingoes were only opportunistically sighted during the study, all sightings occurred in Limeburners Creek NR (unbaited; 3 adults, 2 juveniles). More extensive sand plot data is needed to clarify dingo abundance and activity in each study area.

Strong selection pressure by mesopredators on rodents predicts that anti-predator behaviors should develop in baited areas. Consequently, optimal foraging theory suggests that giving up densities should increase with increased predation risk (Brown 1988). The higher giving up densities observed in baited areas was supportive of our initial predictions. While higher GUDs and small mammal abundance data in baited areas were consistent with mesopredator presence, differences in GUDs may have been due to other factors. Density of rodents was higher for all species in the unbaited areas, which suggests the possibility of multiple encounters of prey per a foraging patch. Consequently, giving up densities would reflect population density rather than the foraging choice of the last individual. Analyzing the number of individuals visiting a patch is difficult to determine from track data. However, none of the cameras placed observed more than one individual at a foraging tray. An additional alternative explanation that has been suggested for lower GUDs is that at higher

rodent population densities, intraspecific competition for food resources results in a lower state of energy of rodents (proposed Brown et al. 1997, Searle et al. 2008, Letnic and Dworjanyn 2011). More extensive study of risk sensitive foraging would be required to assess the validity of this concern. Similarly, risk sensitive foraging based on microhabitat use was not examined. Small mammals may prefer foraging patches placed under cover or may forage more under a new moon rather than full moon (Brown et al. 1997).

Risk foraging was only evaluated in rodents and may differ in medium sized-mammals. The viability and accessibility of mealworm foraging trays have been a factor in the lack of foraging from patches. In future studies, a cover would need to be placed over the worms to prevent overheating or water filling the trays. Palatability by native fauna may be another deterrent as cameras placed at the trays detected an echidna but no mealworms were foraged.

Faunal abundance trends were inconsistent between the visual and scat/trace survey methods. While macropod abundance was higher in baited sites during afternoon and spotlighting surveys, dung transects and sand plots suggested greater macropod presence in nonbaited areas. Other studies of dingo-macropod interactions have found increases in kangaroo abundance at low density of dingoes due to predation (Corbett & Newsome 1987; Thomson 1992; Letnic et al. 2009). The similar large prey densities between baited and nonbaited areas in this study may be reflective of macropod population influx from surrounding cleared land where dingoes are absent due to human exclusion. Macropods were visually observed on private land near Limeburners Creek NR during the study.

Overall, trends in the surveys of medium-sized mammals (bandicoots, potoroos, echidnas, hares, koalas) demonstrated greater density in the nonbaited study area. These findings are consistent with predictions from the mesopredator release hypothesis. Previous scat comparison of fox and dingo diets elsewhere in Australia have shown resource

partitioning. Small and medium-sized prey comprise a greater proportion of fox diets than dingo diets (DECC 2007; Mitchell and Banks 2005). However, glider and possum abundances were unaffected by dingo absence or presence. Arboreal species such as possums and gliders may be less influenced by mesopredation if foraging behaviors shift to minimize time spent on the ground.

Limitations of this study include the difficulties of comparing two spatially segregated habitats. Differences in vegetation between study areas could play a factor in the differences observed. Vegetation density was similar in both study areas; however, canopy cover was greater in the nonbaited sites. A denser canopy could contribute to differences in forest ecosystem communities; nevertheless, understory complexity and height were similar between sites. Previous study on the composition of ground dwelling mammals in NSW eucalypt forests suggests understory complexity and the density of understory shrubs determines ecosystem composition (Catling and Burt 1995). Similarly, small mammal abundance is dependent on regeneration of understory cover after a fire rather than time (Monamy and Fox 2005). While differences in habitat between sites could have influenced the trends observed, understory complexity and height were comparable between study areas and thus less likely to play a determining role in foraging behavior.

While the results of this study provide support for small mammal diversity and abundance under nonbaiting regimes, potential shortcomings include the short time period of the study and lack of replicates. Seasonality effects could play a role in predator-prey population fluctuations, which would not have been captured in our data. Furthermore, differences observed may be locally contained and influences of top order predators may not be consistent in other forest ecosystems. This preliminary study is the first part of a larger project that will examine several other forest ecosystems in New South Wales.

5. Conclusion

The influence of top predators on ecosystem communities through limitation on mesopredators and subsequent trophic cascade effects is gathering support as a consideration in ecosystem management. Invasive mesopredators threaten native wildlife and can result in local prey population extinctions. This study demonstrated the cascade effects of a top predator on foraging behavior of small mammals. With increased mesopredation, prey must compensate for increased predation risk by reducing foraging time. While baiting has been widely used in Australia to control invasive species, if the presence of a top order predator can regulate mesopredation and thus promote small mammal diversity, nonbaiting management of forest ecosystems may be of significant value.

6. References

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