



Assessment of Detection Methods and Vegetation Associations for Introduced Finlayson's Squirrels (*Callosciurus finlaysonii*) in Italy

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Abstract

Managing biological invasions requires rapid, cost-effective assessments of introduced species' occurrence, and a good understanding of the species' vegetation associations. This is particularly true for species that are elusive or may spread rapidly. Finlayson's squirrel (*Callosciurus finlaysonii*) is native to Thailand and southeastern Asia, and two introduced populations occur in peninsular Italy. One of the two introduced populations is rapidly expanding, but neither effective monitoring protocols nor reliable information on vegetation associations are available. To fill this gap, we conducted visual surveys and hair tube sampling in a periurban landscape of southern Italy to compare the effectiveness of these two methods in assessing presence of Finlayson's squirrel. We also determined the species' association with vegetation types at detection locations and nesting sites. Both visual and hair tube sampling effectively assessed the species' presence, but hair tubes resulted in fewer false absences. Moreover, when we controlled for the costs of labor and equipment, hair tubes were 33.1% less expensive than visual sampling. Presence of squirrels and their nests was positively correlated with shrub species richness, indicating that the occurrence of forests with well-developed understory may inhibit the spread of the species.

Keywords *Callosciurus finlaysonii* · Early detection · Hair tubes · Nest selection · Occupancy models

Introduction

Countering the expansion of an introduced animal species, and minimizing its undesirable effects, requires good

knowledge of its distribution and the identification of the areas and vegetation types where expansion may occur (e.g., Ballari et al. 2016). Such information is fundamental to plan eradication or control actions and improve their effectiveness (Braysher 1993; Bertolino et al. 2005). Monitoring newly established populations of introduced species and assessing their vegetation associations is therefore necessary to better understand their spatial dynamics and inform effective management strategies.

Monitoring low-density, elusive, or nocturnal species is challenging (Mills et al. 2000; Thompson 2004; Kindberg et al. 2009; Srivathsa et al. 2014) because false absences may be common, i.e., the species often may not be detected despite being present (Gu and Swihart 2004; Mortelliti and Boitani 2007, 2008). False absences may limit the effectiveness of management actions, which might overlook sites where the species inaccurately is deemed to be absent (Gu and Swihart 2004). For mammals, indirect methods such as detection of footprints (Reynolds et al. 2004; Yarnell et al. 2014) or hair (Gurnell et al. 2004a; Harris et al. 2006) may complement or replace direct observation when the latter is ineffective, for instance when the species avoids areas in which humans are present and possibly is elusive. Moreover, monitoring or countering the spread of introduced

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species greatly depends on the availability of information on their habitat, including nesting or roosting sites (Palmer et al. 2013). For example, reproduction and wintering of introduced squirrels relies on the presence of nesting sites (dreys) (Setoguchi 1991; Okubo et al. 2005; Palmer et al. 2013).

Finlayson's squirrel (*Callosciurus finlaysonii*) is an arboreal rodent native to Southeast Asia (Thorington et al. 2012). The species are ecologically adaptable, and, in its native and introduced ranges, may occur in several forest types. The species occurs in forests that are logged commercially, where it feeds opportunistically and seasonally on fruits, seeds, buds, and occasionally small animals (Lekagul and McNeely 1977; Bertolino et al. 2004). A few populations of this squirrel, introduced through the pet trade (Bertolino and Lurz 2013), have established outside its native range in the last 35–40 years. Two introduced populations are present in Singapore and Japan (Oshida et al. 2007). Two other introduced populations occur in Italy; a small population in the north (Mazzoglio et al. 2007) and a large population in the south resulted from the release, in the 1980s, of a few individuals in an urban park (Aloise and Bertolino 2005). The effects of this species in its introduced range include tree bark stripping, consumption of fruits and seeds in crops, and damage to electric cables (Bertolino et al. 2015; Mori et al. 2016). As in its native range, in its introduced range *C. finlaysonii* is associated with forests, where it builds nests from plants (Bertolino et al. 2004), but no detailed description of habitat is available. Although the population of *C. finlaysonii* in southern Italy has expanded along the Tyrrhenian coast, both naturally and through further human-assisted introductions (Aloise and Bertolino 2005), monitoring has been practically nonexistent (Bertolino et al. 2015).

Effective monitoring requires designing appropriate surveys. Given that *C. finlaysonii* is arboreal and associated with forests, in which direct visual observation may be difficult, indirect detection may be preferred over direct observation. To date, however, no study has assessed the effectiveness of alternative survey methods. Moreover, there is no comprehensive understanding of vegetation associations of *C. finlaysonii*, which would help focus survey efforts and inform future management, including population control or eradication.

We aimed to help fill these gaps. First, we determined whether *C. finlaysonii* were present by applying two detection methods, visual sampling (VS) and hair tube sampling (HTS), both of which are used to survey European red squirrels *Sciurus vulgaris* (Mortelliti and Boitani 2008). We compared the effectiveness and cost of these methods. Then, we assessed vegetation associations of *C. finlaysonii* at two levels: forest patches and nesting sites.

Methods

Patterns of Patch Occupancy and Comparison of Detection Methods

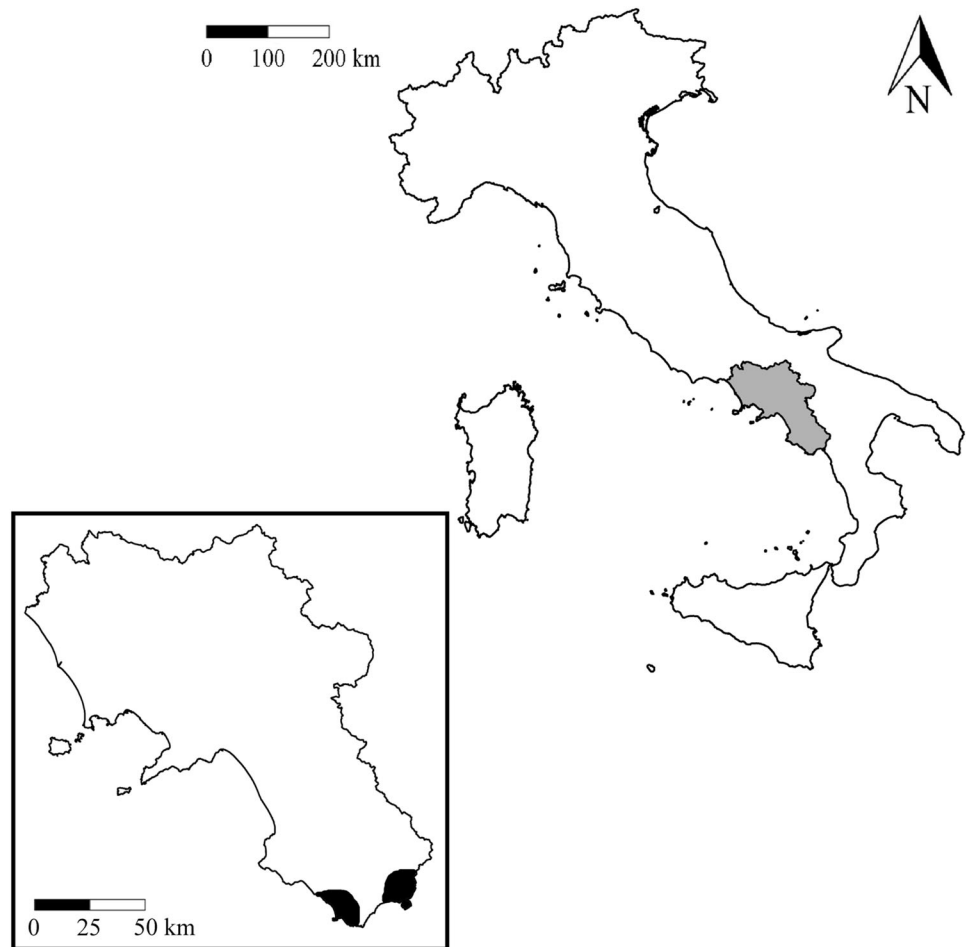
We used both VS and HTS to survey 22 forest patches (mean size \pm SD: 4.3 ± 0.9 ha, range 0.6–18.1 ha) within a periurban landscape in the municipalities of Sapri (40.07°N, 15.64°E) and Palinuro (40.03°N, 15.28°E) on the Tyrrhenian coast of Campania (southern Italy; Fig. 1), where Finlayson's squirrels were first recorded in 2004 (Aloise and Bertolino 2005). Forest patches varied in size, isolation, and species composition (Table 1), and were surrounded by an agricultural matrix of olive groves, cereal fields, and private vegetable gardens and by urban settlements.

Hair tubes were commercially available PVC pipe segments (length 30 cm, diameter ~7 cm), at each end of which a small piece of plastic sheet covered with sticky tape was fixed to the roof (Gurnell et al. 2004a). We baited hair tubes with hazelnut cream and sunflower seeds and secured tubes horizontally on tree trunks and branches 1.5–2.0 m above ground. We established one transect per forest patch, each consisting of a linear sequence of hair tubes ca. 100 m apart. The number of hair tubes in each transect increased as patch size increased, and ranged from 2 to 12 (Mortelliti and Boitani 2008); for forest patches <1.5 ha, we positioned only one tube. We inspected hair tubes on 8 occasions separated by 10 days (80 days total). During each inspection, we replaced adhesive tape and stored trapped hairs for subsequent laboratory identification. Hairs were slide-mounted and identified with an Olympus BX 51 optic microscope equipped with 4–100 \times lenses. We examined hair morphology following Teerink (1991) and Venturini et al. (2008), and used a reference collection. We used this approach to distinguish *C. finlaysonii* hairs from those of other rodents that might visit hair tubes (*Glis glis* [edible dormouse], *Muscardinus avellanarius* [hazel dormouse], and *Rattus rattus* [black rat]). No other squirrel species occurred in the area.

For VS, one operator walked along a given transect, stopping 5 min near each tube and recording all squirrels sighted (Gurnell et al. 2004a). We therefore recorded squirrels on the basis of point counts, without accounting for their distance from the observer. VS also was conducted every 10 days for a total of 8 visits.

For each patch, we compiled the detection history (non-detection, 0; detection, 1) for HTS and VS separately. We then ran single-season occupancy models (Mortelliti and Boitani 2008), including a set of environmental covariates (Table 1) likely to be associated with patterns of occupancy in arboreal squirrels in a patchy landscape (Mortelliti and Boitani 2008). We expected the probability of presence in a given patch (Ψ : psi) to be associated with patch size, degree

Fig. 1 Study region (Campania, shaded area) and sampling sites (inset; shaded areas in circles)



of isolation as measured by either the number of patches within 500 m of the patch edge or the distance between the focal patch and the closest neighboring patch, and vegetation structure. We selected the 500 m as the maximum distance that other squirrel species are known to move across a forest gap, representing a proxy for dispersal ability in a matrix of non-habitat (Bakker and Van Vuren 2004). We classified forest composition as mixed oaks, mono-specific cork oak, or mixed (i.e., coniferous and broad-leaved) forest. We measured shrub species richness as the number of shrub species recorded along each 10 m transect or, in two patches <1.5 ha large where only one hair tube was present, in a 10 m circular area around the tube. Because introduced *C. finlaysonii* often occur close to urban areas (Aloise and Bertolino 2005), we also included the minimum distance from the closest urban settlement as a covariate. We modeled detection probability (p) as a function of patch area, shrub species richness, and forest composition to account for potential biases in squirrel detection resulting from differences in the area within which transects were embedded or in station-dependent hair tube attractiveness or visibility.

We ran analyses with the R package *Unmarked* (Fiske and Chandler 2011). We evaluated sample size, i.e., the minimum number of visits necessary to assess whether the species was present (Reed 1996), as $N = \ln(\alpha\text{level})/\ln(1-p)$, where α represents the probability of type I error (fixed at 0.05). We ranked models according to Akaike Information Criterion (AIC) values (Burnham and Anderson 2002). Because different models may provide similar results, we considered as valid all models with $\Delta\text{AIC} < 2$ from the model with the lowest AIC (Burnham and Anderson 2002). Among valid models, we assumed that the best supported were those with the lowest AIC values (MacKenzie et al. 2006). We calculated Akaike weights (w_i) to assess the strength of evidence in support of each model (Srivathsa et al. 2014). We also used a stepwise process, implemented with the *lme4* R package (Bates et al. 2015; R Core Team 2014), to generate a global model of Ψ and identify the covariates that were significantly associated with occupancy.

We also estimated the cost of implementing each survey method for the entire study period and per survey. Costs included materials (hair tubes, bait) and human effort (the

Table 1 Characteristics of 22 forest patches in southern Italy in which we used hair tube and visual sampling to establish whether *C. finlaysonii* were present

| Patch area (ha) | Forest type ^a | Number of patches within 500 m of the focal patch | Distance to closest patch (m) | Shrub species richness | Distance from urban sites (m) |
|-----------------|--------------------------|---------------------------------------------------|-------------------------------|------------------------|-------------------------------|
| 1.1 | Mixed | 2 | 60 | 2 | 170 |
| 2.6 | Mixed oaks | 2 | 60 | 2 | 360 |
| 15.2 | Cork oak | 2 | 110 | 1 | 720 |
| 4.5 | Mixed oaks | 0 | 950 | 4 | 230 |
| 1.9 | Mixed | 0 | 1300 | 2 | 120 |
| 3.3 | Conifer | 0 | 1450 | 1 | 680 |
| 13.1 | Conifer | 0 | 1450 | 1 | 450 |
| 1.5 | Mixed | 0 | 750 | 2 | 390 |
| 2.3 | Mixed oaks | 1 | 320 | 2 | 670 |
| 0.6 | Mixed oaks | 2 | 330 | 3 | 1090 |
| 2.3 | Cork oak | 2 | 360 | 4 | 750 |
| 1.7 | Mixed | 1 | 340 | 4 | 1570 |
| 6.3 | Mixed oaks | 1 | 340 | 2 | 1060 |
| 1.2 | Mixed | 0 | 970 | 1 | 1340 |
| 3.1 | Mixed | 1 | 460 | 2 | 180 |
| 4.1 | Mixed oaks | 0 | 550 | 4 | 240 |
| 11.2 | Cork oak | 1 | 60 | 3 | 190 |
| 2.7 | Mixed oaks | 0 | 660 | 3 | 360 |
| 3.5 | Mixed oaks | 1 | 250 | 2 | 720 |
| 3.7 | Mixed | 1 | 250 | 2 | 480 |
| 18.1 | Conifer | 0 | 1500 | 1 | 1205 |
| 4.2 | Mixed oaks | 1 | 60 | 2 | 320 |

^aMixed: coniferous and broadleaved forest; mixed oaks: mixed broadleaved forest with two or more oak (*Quercus* spp.) species

time needed to assemble and place hair tubes and to commute between tubes and between sites, and time spent in the field and in the laboratory, at a standard rate of 10.00 €/h). We calculated total cost by multiplying cost per visit by the minimum number of visits needed to assess occupancy. We assessed cost per detection by dividing the total cost by the number of detections with the VS and HTS methods.

Selection of Nesting Areas

C. finlaysonii builds conspicuous nests of leaves and twigs on tree trunks, branches, or in the foliage (Setoguchi 1991; Okubo et al. 2005). We located squirrel nests by inspecting forest patches along all accessible trails separated by ca. 50 m, following the approach adopted by Palmer et al. (2013) for red-bellied squirrels *Sciurus aureogaster* introduced to

Florida, and recorded their position with a Dakota 10—Garmin GPS receiver.

We measured nest site characteristics at three levels: nest, nesting tree, and plot. At the nest level, we measured height above ground and nest exposure (measured with a compass as the orientation of the nest relative to its support, which we classified as one of the eight main cardinal directions). The position and exposure of the nest may affect the timing and duration of solar radiation to the nest, influencing microclimate within the nest. We explored differences in nest aspect with a Rayleigh test, implemented in the Circ-Stat R package (Agostinelli 2009).

For each tree in which a squirrel nest was built, we recorded tree species, tree height and diameter at breast height (DBH), whether vines were present on the trunk, percent canopy closure (assessed visually following Paletto

and Tosi 2009), number of crown linkages (i.e., number of neighboring trees whose branches were ≤ 0.5 m from the occupied tree), location in the patch (core or edge of patch or clearing), and distance from the closest tree. We also generated 50 random coordinates within the study area with a random number grid on 1:10,000 maps. We used the tree closest to each random location as a control tree, at which we took the same measurements that we recorded for nesting trees. At the random location, we chose the closest tree with DBH > 10 cm, the lowest value in the nesting trees data. To establish which trees are used for nesting and which features are associated with those trees, we compared the features of used trees with those of unused (random) trees. We used logistic regressions, including both categorical variables (tree species, whether the tree was live or dead, presence of vines, tree position within the forest patch) and normally distributed, continuous variables (tree size, canopy closure, and number of linkages).

At the plot level, we collected vegetation data within 10 m of nesting and random trees (Palmer et al. 2013). We recorded shrub species richness and abundance, tree species richness, and densities of live and dead trees (Pignatti 1982). We explored differences in vegetation between plots in which the species nested and random plots by running generalized linear models. We evaluated the direction and magnitude of effects by inspecting parameter estimates (β).

We ran all analyses with R 3.2.1 and set significance at $p < 0.05$. We present all results as mean \pm SE with associated 95% confidence intervals (CI).

Results

Patterns of Patch Occupancy and Comparison of Detection Methods

We detected *C. finlaysonii* in 19 of 22 forest patches (86.4%). The proportion of occupied sites estimated by HTS was closer to the observed proportion than that estimated by VS (HTS: range: 83–87%, CI: 0.84–0.85; VS: 70–83%, CI: 0.72–0.80). VS and HTS resulted in 17 and 4% false absences, respectively. Detection probability (p) estimated with either method was similar (range 0.45–0.58), although it was significantly higher for HTS (CI: 0.56–0.58) than for VS (CI: 48.4–53.6; see Table 2). For both methods, the minimum number of visits per site necessary to establish presence was 5. No covariates were significantly associated with detection probability. In the global model, Ψ was positively correlated with shrub species richness and, to a lesser extent, negatively correlated with the distance to the closest patch; patch area and forest type were associated with occupancy as estimated by VS and HTS, respectively (Table 2).

Table 2 Variables associated with occupancy by *C. finlaysonii* as estimated by visual ($R^2 = 0.89$) and hair tube ($R^2 = 0.86$) sampling, respectively

| | Variable | β | SE | p |
|-----------------|---------------------------------|---------|-------|--------|
| Visual sampling | Shrub species richness | 0.243 | 0.067 | <0.001 |
| | Distance to closest patch | -0.125 | 0.061 | <0.05 |
| | Forest type | 0.688 | 0.089 | <0.05 |
| | Distance from human settlements | -0.673 | 0.046 | <0.05 |
| | Patch area | 0.079 | 0.054 | 0.12 |
| | Intercept | 0.907 | 0.117 | <0.001 |
| Hair tubes | Shrub species richness | 0.467 | 0.076 | <0.001 |
| | Patch area | 0.104 | 0.056 | <0.05 |
| | Distance to closest patch | -0.116 | 0.061 | <0.05 |
| | Distance from human settlements | -0.098 | 0.012 | 0.08 |
| | Forest type | 0.083 | 0.090 | 0.47 |
| | Intercept | 1.026 | 0.030 | <0.001 |

SE standard error

Table 3 Estimated total costs of visual and hair tube sampling to assess occupancy of introduced *C. finlaysonii* in 22 forest fragments in southern Italy (hourly salary: 10.00€)

| | Hair tube sampling | Visual sampling |
|----------------------------|--------------------|-----------------|
| Materials | 130.00€ | – |
| Assemblage and positioning | 230.00€ | – |
| Time in the field | 3187.50€ | 3231.75€ |
| Time in the laboratory | 150.00€ | – |
| Total cost | 3697.50€ | 3231.75€ |
| Number of detections | 63 | 36 |
| Cost per detection | 58.69€ | 87.67€ |

The two methods had similar costs (Table 3), 2984€ and 2585€ for HTS and VS, respectively (Table 3), but after correcting for the number of surveys, HTS was 33.1% less expensive than VS (58.69€ vs. 87.67€ per detection). We provide a full list of the models we evaluated as Supplementary Material (Tables S1 and S2).

The models that minimized information loss (lowest AICc) and maximized explained proportion of variance (R^2 values) (Table 4) indicated that occupancy was a function of shrub species richness when estimated with HTS, and a function of shrub species richness and distance from other forest patches when measured with VS.

Occupancy was positively associated with shrub species richness (0.24 and 0.47 for VS and HTS, respectively) and negatively associated with distance from other patches (-0.13 and -0.11; Table 2). VS suggested that the likelihood of squirrel occupancy significantly increased near human settlements (-0.67; Table 3). Forest type also was

Table 4 Models of *C. finlaysonii* occupancy in fragmented forest in southern Italy

| | AIC | Δ AIC | w | Ψ | p | N |
|----------------------------------------------------------------------|--------|--------------|------|--------|------|-----|
| Model—Visual sampling | | | | | | |
| Ψ (Distance to closest patch + Shrub richness) $p(\cdot)$ | 75.59 | 0.00 | 0.21 | 0.83 | 0.54 | 5 |
| Ψ (Forest type + Shrub richness) $p(\cdot)$ | 76.40 | 0.81 | 0.14 | 0.70 | 0.52 | 5 |
| Ψ (Patch Area + Shrub richness) $p(\cdot)$ | 76.46 | 0.86 | 0.14 | 0.73 | 0.53 | 5 |
| Ψ (Shrub richness) $p(\cdot)$ | 76.82 | 1.23 | 0.11 | 0.73 | 0.49 | 5 |
| Ψ (Distance from human settlements + Shrub richness) $p(\cdot)$ | 77.22 | 1.63 | 0.09 | 0.80 | 0.45 | 5 |
| Model—Hair tubes | | | | | | |
| Ψ (Shrub richness) $p(\cdot)$ | 166.36 | 0.00 | 0.20 | 0.85 | 0.58 | 5 |
| Ψ (Patch Area + Shrub richness) $p(\cdot)$ | 166.47 | 0.11 | 0.19 | 0.87 | 0.57 | 5 |
| Ψ (Distance to closest patch + Shrub richness) $p(\cdot)$ | 167.15 | 0.79 | 0.14 | 0.83 | 0.56 | 5 |
| Ψ (N patches in 500 m + Shrub Richness) $p(\cdot)$ | 167.92 | 1.56 | 0.09 | 0.83 | 0.58 | 5 |
| Ψ (Forest type + Shrub richness) $p(\cdot)$ | 168.07 | 1.71 | 0.08 | 0.84 | 0.57 | 5 |
| Ψ (Shrub richness + Distance from human settlements) $p(\cdot)$ | 168.32 | 1.96 | 0.08 | 0.84 | 0.58 | 5 |

AIC Akaike Information Criterion, w model weight, Ψ occupancy, p detection probability, N minimum number of visits to assess species presence

associated with occupancy by *C. finlaysonii* (0.688 and 0.083 for HTS and VS, respectively), with lowest values in conifer and cork oak stands, and highest values in mixed and mixed oak patches.

Selection of Nesting Areas

The 47 nests were located at heights of 7.7 ± 0.4 m above ground (range: 2.0–20.5 m), often (48.9%) between 5 and 6 m. Nests were located in all patches that VS and HTS indicated were occupied. Nests faced all directions, but most often southeast (Rayleigh's $Z = 123.81$, $p < 0.001$, $r = 0.32$). Most ($n = 33$) nests were located in oaks (*Quercus* spp.). The remainder were located in strawberry trees (*Arbutus unedo*) ($n = 7$), olive trees (*Olea europaea*) ($n = 3$), Spanish broom (*Spartium junceum*) ($n = 2$), and wild plum trees (*Prunus* spp.) ($n = 2$). Nests occurred most often in downy oaks (*Quercus pubescens*) ($\beta = 1.43$, $p < 0.05$) and least often in cork oaks (*Q. suber*) ($\beta = -2.46$, $p < 0.05$).

The distance between random trees and closest nesting trees was 72–358 m. The size of nesting trees did not differ from that of random trees (height: $\beta = 1.11$, $p = 0.11$, CI: -1.59 – 5.0 ; DBH: $\beta = 0.91$, $p = 0.061$, CI: -1.64 – 1.96). Presence of vines on the trunk ($\beta = -0.15$, $p = 0.54$, CI: -1.11 – 3.12), tree position within the patch ($\beta = 0.35$, $p = 0.91$, CI: -2.12 – 3.21), canopy closure ($\beta = -0.55$, $p = 0.78$, CI: -9.9 – 15.4), and distance from the closest tree ($\beta = 0.29$, $p = 0.13$, CI: -1.21 – 3.39) did not vary between nest trees and random trees. Nesting trees had fewer crown linkages than random trees (nest trees: 3.6 ± 0.40 ; random trees: 5.3 ± 0.1 ; $\beta = -3.19$, $p < 0.05$, CI: -5.12 to -1.09). Plots around nesting trees had a lower abundance of shrubs (nesting plots: 20.70 ± 1.01 ; random plots: 35.44 ± 1.51 ;

$\beta = -2.78$, $p < 0.01$, CI: -8.12 to -0.91) but greater species richness (nesting plots: 8.54 ± 2.38 ; random plots: 3.26 ± 1.20 ; $\beta = 4.31$, $p < 0.01$, CI: 1.12 – 6.33) than plots around random trees. Neither tree species richness ($\beta = 1.91$, $p = 0.34$; CI: -0.91 – 1.43) nor tree density ($\beta = 0.36$, $p = 0.71$, CI: -0.12 – 0.61) was significantly different between nesting and random plots.

Discussion

We found that HTS is more effective and costs less than VS to detect *C. finlaysonii* (Cagnacci et al. 2012). Our cost analysis did not take into account the fact that plastic hair tubes are robust enough to be reused for several surveys, further reducing costs. The proportion of sites in which we detected the species by HTS was ~4% lower than the estimated occupancy. Our results suggested that in forests, a minimum of five visits, 10 days apart, will provide reliable results. Although we conducted eight hair trapping surveys, we did not detect squirrels in additional sites after the fifth visit. The two methods differ markedly in sampling time. Whereas hair tubes typically are deployed over weeks or months, VS is conducted over short durations. VS may be affected by squirrels' antipredatory responses, which include escaping or hiding from the observer (Haigh et al. 2017). When threatened, *C. finlaysonii* and congeners (Tamura and Yong 1993) emit loud alarm calls that increase detection probability (personal observation).

The positive association between occurrence and shrub species richness at both tree and plot levels might reflect the species' opportunistic foraging on a range of plant parts, particularly seeds and fruits that are available on shrubs year round (Bertolino et al. 2004). The dependence of

C. finlaysonii on large and well-connected forests, which is typical of tree squirrels (Celada et al. 1994; Rodríguez and Andrés 1999; Koprowski 2005), may explain the negative association between occupancy and distance to neighboring forest patches. The presence of *C. finlaysonii*, like that of other adaptable species often introduced to urban areas by the pet trade, is positively associated with proximity to human settlements (Jodice and Humphrey 1992; McCleery et al. 2007; Bonnington et al. 2014; Greene and McCleery 2017).

Although the diet of *C. finlaysonii* mainly consists of ripe fruits (Bertolino et al. 2004), in the study area these were available only for part of the year. Therefore, nesting on *Q. pubescens* may be a strategy to facilitate harvesting of acorns, providing squirrels with an additional food source, when other food items such as berries are unavailable. Southeast-facing dreys may facilitate exposure to the sun in the early morning while reducing exposure to cold wind from the north, thus providing a warm microclimate to nestlings. This may also explain the low heights of most nests. Most medium-sized tree squirrels (i.e., 100–500 g) build their nests at a similar height above ground (ca. 5 m) to shelter the nest from strong wind while precluding access to terrestrial predators or competitors that might steal the food stored in the nest (Rothwell 1979; Alvarenga and Talamoni 2005; Thorington et al. 2012). Given the tropical-subtropical origin of these species (Lekagul and McNeely 1977; Thorington et al. 2012), warm nesting sites also may be an adaptation to the cold spells in temperate regions (Terrien et al. 2011).

Our work highlights that hair tubes may be an effective means of surveying *C. finlaysonii* in dense vegetation or where the species may occur at low densities, e.g., where it might have colonized recently. We found that *C. finlaysonii* often occurs and builds nests in forest patches in which the species richness of the understory is relatively high. Such patches may be high priorities for monitoring and for management actions, such as control or eradication.

Introduced squirrels have had considerable effects in European ecosystems (Bertolino 2009; Bertolino and Lurz 2013). For example, native *Sciurus vulgaris* (Eurasian red squirrels) have been driven to extinction in much of their range within the United Kingdom by *Sciurus carolinensis* (Eastern gray squirrels) (Gurnell et al. 2004b) and, in Italy, their survival is jeopardized by both *S. carolinensis* (Bertolino et al. 2014b) and *Callosciurus erythraeus* (Pallas' squirrels) (Mazzamuto et al. 2016). *C. finlaysonii* removes bark from trees, causing damage to forest and orchards (Mori et al. 2016; Bertolino et al. 2015), and may have negative effects on other species. The effects of introduced squirrels on other species and human activities may make active monitoring of their range useful for predicting and informing management of their expansion at larger extents

(Di Febbraro et al. 2016; White et al. 2016). *C. finlaysonii* is believed to be expanding south, where it may compete with the newly described *Sciurus meridionalis* (Calabrian black squirrel) (Bertolino et al. 2014a), an endemic species occupying a small area (Wauters et al. 2017). We therefore urge that monitoring of the southern Italian population of *C. finlaysonii* be carried out to inform the adoption of effective control measures.

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Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no conflict of interest.

Ethical Approval All procedures performed in studies involving wild species were in accordance with the ethical standards of the Institutional or National Research Committee and with approved ethical standards.

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