Distribution and abundance of hollow-bearing trees in urban forest fragments.

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Abstract

Habitat structures, such as hollow-bearing (i.e. cavity-bearing) trees, are globally recognised as important forest features for wildlife conservation and for providing important structural heterogeneity in natural and modified landscapes. The depletion of structural resources, such as hollow-bearing trees, within a landscape, may therefore be limiting to biota where no other functional substitutes exist. We surveyed 45 natural forest remnants across the rapidly urbanising City of Gold Coast, south-east Queensland, Australia to quantify the distribution and abundance of hollow-bearing trees. Tree and tree-hollow variables were quantified within the selected 91 plots. In total 6048 trees from 34 eucalypt species were sampled and 916 hollow-bearing trees containing 2159 hollows were recorded. The average hollow-bearing tree density ($37.5 \pm 3.13/\text{ha}$) was much higher than those found in other studies within Australia. Hollows were more prevalent in large trees and the majority (50.4%) of hollows were bayonets (10 cm). Dead and decaying trees also had a greater likelihood of having a hollow than healthy trees. Our results highlight the value of urban forest remnants in maintaining the functional capacity of urban landscapes for biodiversity by protecting hollow-bearing trees as habitat resources. This information will assist conservation managers and planners to establish sound adaptive management plans to conserve biodiversity and ecosystem function in natural and modified landscapes.

Keywords: Urbanisation, forest patch, habitat structure.

46 **Introduction**

47 The destruction and removal of natural habitats is proceeding at a rapid rate globally
 48 (Rankmore and Price, 2004; Lewis et al., 2009; Gouldie, 2013). Within Australia, the
 49 clearing and disturbance of natural vegetation and the subsequent habitat fragmentation, are
 50 recognised as principal drivers behind biodiversity declines (Williams et al., 2001; Bradshaw,
 51 2012). Australian forests have been severely modified since European settlement (Wells et
 52 al., 1984; Lunney, 2004), with over 80% of remaining eucalypt forests having been subjected
 53 to anthropogenic alteration. (Norton 1996). The combined effects of forest loss and
 54 degradation have resulted in severely fragmented landscapes (Bradshaw, 2012). However,
 55 within a fragmented landscape, small and often isolated forest patches are valuable to
 56 conservation, as they represent former habitat that was once common in any given area (van
 57 der Ree et al., 2003). Forest patches are therefore important functional elements within larger
 58 landscapes encompassing the urban matrix. Within urbanised regions the transformed matrix
 59 (e.g. residential areas) may be considerably less permeable than contiguous habitat for some
 60 species and is known to reduce ecosystem function and biodiversity in general (Ewers and
 61 Didham, 2006). Therefore, the retention of natural habitats as refugia becomes increasingly
 62 important in these regions (Harper et al., 2005a).

63
 64 Within urban landscapes refugia are found in forest patches as well as urban parks and
 65 gardens, with their value depending on their ability to provide resources for indigenous flora
 66 and fauna (Godefroid and Koedam, 2003; Harper et al., 2005a; Alvey, 2006). The
 67 availability of one resource, tree hollows, generally limits the distribution and abundance of
 68 many obligate hollow-bearing tree (HBT) users and general species richness of an area
 69 (Gibbons and Lindenmayer, 2002). The importance of HBTs for a large range of fauna has
 70 been demonstrated globally (Bai et al., 2003; Aitken and Martin, 2004; Monterrubio-Rico

71 and Escalante-Pliego, 2006; Holloway et al., 2007; Blakely et al., 2008; Goldingay, 2009).
72 Hollow-bearing trees are particularly important for Australian faunal communities, given the
73 large number of species (approximately 300) which are reliant upon them as habitat trees
74 (Gibbons and Lindenmayer, 2002; Goldingay, 2009, 2011).
75
76 HBTs are a highly valued resource within the Australian context (Maloney et al., 2002;
77 Wormington et al., 2003; Goldingay, 2011). There is, however, a general paucity of baseline
78 data surrounding the availability of habitat structures in urban landscapes (Le Roux et al.,
79 2014a). It is therefore important to determine the ability of forest patches to provide
80 functional refuges for biodiversity against the backdrop of urban development. Furthering
81 our understanding of the availability of this resource within the urban landscapes would
82 provide the foundation for a more holistic appraisal of biodiversity facets in urban planning.
83 Previous studies have investigated the impacts of urbanisation on the distribution and
84 abundance of hollow-bearing trees (Harper et al., 2005a; van der Ree and McCarthy, 2005;
85 Harper et al., 2008), and more recently, the persistence of these structures in urban landscapes
86 (Le Roux et al., 2014a; 2014b). Loss of these habitat structures also has cascading effects on
87 urban biodiversity more generally (Harper et al., 2005a). The retention of hollow-bearing
88 trees is therefore vital to the maintenance of ecosystem integrity in urban environments. This
89 paper aims to quantify the distribution and abundance of hollow-bearing trees in a rapidly
90 urbanising region of Australia by assessing their availability within the landscape in habitat
91 remnants. We expand on the results and discuss options for management prescriptions for
92 the retention of HBTs within remnant forest patches in urban landscapes.

93

94 **Methods**

95 This study was completed on the Gold Coast, eastern Australia. The Gold Coast is
 96 Australia's sixth largest city with over 530 000 residents and a significant transient tourist
 97 population of more than 12 million each year. Natural habitats comprise approximately 50%
 98 of the Gold Coast covering ~63 600 ha (Gold Coast City Council, 2009). Rapid and ongoing
 99 urbanisation, however, continues to erode and fragment these natural areas, particularly in
 100 urban and peri-urban areas (Spearbitt, 2009; Pickering et al., 2010).

101

102 To assess the status of HBTs on the Gold Coast we surveyed natural forest patches within wet
 103 and dry sclerophyll vegetation communities that dominate this region. Replicate forest
 104 patches that were each greater than 2 ha and below 500m elevation (to exclude rainforest
 105 communities) were identified from within council Conservation Areas, National Parks, State
 106 Forests and on private land. Of the 74 sites that fulfilled the selection criteria above, 45 were
 107 finally used for this study and these ranged from the highly developed coastal plains through
 108 to the more natural environments of the hinterland (Fig. 1). Sites that were removed were
 109 either inaccessible (i.e. on private land), were not spatially independent, or were substantially
 110 altered through either grazing or mowing and were therefore not representative of an intact
 111 forest environment.

112

113 **Figure 1**

114

115 Five large contiguous forest patches (> 500 ha) served as control sites, three of which were
 116 located in peri-urban areas and two within urban areas. Sampling plots within each forest

patch were defined by placing a grid over all sites, such that the number of plots surveyed was determined by patch size using a hierarchical grid cell size. Grid cell sizes ranged from 250 m in small (2-25ha) forest patches to 2000 m in the largest site (~1700ha). In total, 91 plots were identified from the 45 sites. Sites varied in size, shape, topographic location as well as position along an urban gradient. At each plot a fixed area methodology was used (*sensu* Gibbons and Lindenmayer, 2002), to sample relevant tree community parameters.

Plot sampling protocol

In each plot all eucalypt trees within a 30 m radius from the plot midpoint and with a diameter at breast height over bark (dbh) ≥ 10 cm were identified and measured to determine vegetation community composition and to quantify the status of hollow-bearing trees. Eucalypts (Family: Myrtaceae) were consciously selected as these species typically form hollows over time. Five key genera; *Angophora*, *Corymbia*, *Eucalyptus*, *Lophostemon* and *Syncarpia* were found across all sites, although, not all sites contained all genera. Eucalypt identification followed Brooker and Kleinig (2004) and Euclid (Centre for Plant Biodiversity, 2006), on the basis of bark, fruit bud and leaf characteristics. The area of each circular plot was 0.28 ha resulting in a total survey area of 25.4 ha over the 91 plots.

Hollow-bearing trees detected on each plot were categorised into one of eight senescence types (Fig. 2). Senescence types used here followed that of Whitford (2002), but the number of categories were reduced from 11 to 8; S1 is a completely healthy tree while, S2-S8 represent increasing degrees of senescence, similar to Whitford (2002). The classification of trees based upon the extent of living structures is warranted given the higher value of living trees to fauna than dead trees (Moloney et al., 2002). Tree-hollows in each tree were assessed using binoculars and recorded as one of seven hollow forms (Fig. 3). A 10 cm

minimum entrance diameter threshold was selected based on previous studies that found tree hollows with entrances of this size were large enough to accommodate the larger species of gliders such as *Petaurus australis* (Shaw, 1791) and *Petauroides volans* (Kerr, 1792) (Eyre, 2005). While vertebrate fauna do utilise hollows <10 cm (Goldingay, 2011), these hollows are not easily detectable using ground based surveys (Harper et al., 2005a; Koch, 2008). It is also possible to overestimate the number of tree hollows due to the fact that a tree will often contain many openings which are blind and not of a suitable depth for occupation (Lindenmayer et al., 1990). One way to counter this problem would be to fell trees (Gibbons and Lindenmayer, 2002) or undertake double sampling in the form of tree climbing (Harper et al., 2004). The first option was not considered appropriate or possible for this study, while time and financial constraints made the latter option infeasible. Despite the potential bias in estimating hollow presence in trees, ground based surveys are widely used in a number of different Australian environments (Harper et al., 2004; Eyre, 2005; Koch, 2008).

Figures 2 and 3

The diameter of each hollow entrance was estimated to the nearest 5 cm, using 10 cm as the minimum size. Fissure hollow size estimates were only based on the width (smallest dimension) of each fissure and used the same 10 cm threshold. Tree height and hollow height were measured using a rangefinder (True Pulse 200, Laser Technology Inc.). Hollow-bearing tree density D (number of trees / ha) per site was calculated as the sum of all hollow-bearing trees $HBTs$ from all plots (p) at a site divided by the sum of the total plot area A_p surveyed at each site (Equation 1).

$$D = \frac{\sum_{p=1}^n HBT_p}{\sum_{p=1}^n A_p} \quad (1)$$

Data analysis

Predictor variables known to have a strong biological influence on tree hollows were analysed based on *a priori* investigations. Spearman rank correlation measures were used to identify correlated variables using BIOENV in Primer E version 6.1.11 (Clarke and Warwick, 2001). A correlation coefficient value $|r| > 0.7$ was chosen to identify pairs of highly correlated variables (Garden et al., 2010). BIOENV uses all of the available environmental variables to find the combination that ‘best explains’ the patterns in the biological data (Clark and Ainsworth, 1993). No explanatory variables were strongly correlated and thus all were retained. The decision to retain or remove variables was also based on their known or perceived ecological significance. Significance levels were set at 0.05 for all analyses. No transformations were made to the dependent or predictor variables.

A standard linear regression was used to examine the relationship between tree dbh and the number of hollows it contained. An independent students t-test was used to determine whether the dbh of trees with and without hollows was different. A single factor ANOVA with Fisher’s Least Significant Difference (LSD) post-hoc analysis was used to compare the variation between hollow type categories and hollow size. The presence of a hollow in a tree was tested against the predictor variables of tree form (categorical), dbh and tree height (continuous covariates) using a binomial linear regression model with logit link function. Statistical analyses were conducted using STATISTICA (Statsoft, 2005).

Results

A total of 6048 trees from 34 eucalypt species was recorded from the 45 sites (91 plots) (Table 2). Seventeen species accounted for > 95% of all trees, with five species *Lophostemon confertus*, *Corymbia intermedia*, *Eucalyptus carnea*, *E. crebra* and *E.*

propinqua accounting for 51% of the woody community in the urban forest patches.

Corymbia intermedia was the most common species and was found at 98% of all sites. A total of 916 (14.9% of all trees) HBTs were found from sites ranging from 1 – 86 HBTs per site (mean = 20.4 ± 2.48). Mean HBT density was found to be 37.5 ± 3.13 trees per hectare across all sites (Table 3). These HBTs comprised 25 eucalypt species, as well as unidentified dead trees (Table 2), and contained 2159 hollows (mean = 48.0 ± 6.59) (Table 3). Each site contained an average of six species (range 1-13) of HBTs (Table 3), dominated by *L. confertus* (142) followed by standing dead trees (124) and *E. crebra* (91) (Table 2). All dead trees (form 7 and 8) contained hollows whereas the proportion of living trees (forms 1 - 6) with hollows ranged from 0.01-0.45 (Table 2). The likelihood of trees having a hollow increased significantly as trees senesced (Wald Statistic 451.7, d.f. = 7, $p < 0.01$). Standing dead trees had the most hollows (16%) followed by *L. confertus* (11%) and *E. propinqua* (10.1%) (Table 4).

Tables 2, 3 and 4

The dbh of over 69% of HBTs ranged between 30-60 cm (mean 49.3 ± 0.75 cm), while the dbh of 78% of non-hollow-bearing trees ranged between 20-40 cm (mean 26.8 ± 0.18 cm). The dbh for all trees ranged between 10-159 cm (mean 30 ± 0.22 cm) (Fig 4). The difference between the dbh of trees with hollows and without hollows was significant ($t = -28.75$; d.f. = 879; $p < 0.01$). Trees with a larger dbh also had more hollows ($R^2 0.137$; $p < 0.05$). Hollows within the 10 cm size category were the most abundant comprising 50.4% of all hollows detected (Fig. 5). The relationship between hollow size and type was also significant ($F_{6,2136} = 142.5$; $p < 0.01$), with all hollow types showing substantial variation in hollow size (Fig.

6.). Trunk top hollows were larger than all other hollow types, and bayonets were smaller than trunk-main, branch-main, butt and branch end hollows.

Figures 4, 5 and 6

Discussion

The distribution and abundance of habitat features is an important driver of community diversity (Ficetola and De Bernard, 2004; Ewers and Didham, 2006), particularly in areas where species are heavily dependent on structurally complex features (Goldingay, 2011). Previous studies have recorded a number of hollow-dependent fauna from all vertebrate taxa in urban landscapes (Harper et al., 2005a; Durant et al., 2009; Davie et al., 2013; Treby 2013). This highlights the pivotal importance of HBTs in urban forest patches for the maintaining urban biodiversity. This study found that the density of HBTs across all sites averaged approximately 37.5 HBTs /ha but ranged from 1-86 HBTs /ha. This figure is within the range of HBTs found in tropical environments (Ball et al., 2011), but considerably higher than the minimum number (4-6 /ha) recommended for managed forests in sub-tropical south-east Queensland (Lamb et al., 1998). This figure is also higher than densities of HBTs (range 5.7 – 13.4 HBTs/ha) recorded from other urban environments in Australian temperate regions (Harper et al., 2005a; Le Roux et al., 2014b). The average density of HBTs found in the current study may, however, overestimate the availability of hollows, particularly in terms of their use by fauna, as bayonet hollows (10 cm) accounted for 50.4% of all hollows found. While small hollows are a valuable resource (Eyre, 2005), hollows with a diameter >10 cm are necessary to meet the requirements of larger hollow-using species (Murphy et al., 2003).

Of the 916 HBTs found, standing dead trees represented 13.5% ($n = 124$), which is substantially lower than the 50% reported for production forests in south-east Queensland (Moloney et al., 2002). The high representation of standing dead trees found during that particular study was likely to have been in response to extensive silvicultural practices including poisoning and ringbarking carried out in those forests (Moloney et al., 2002). In general, standing dead trees have a higher likelihood of containing hollows than live trees (Eyre, 2005; Harper et al., 2005a; Beyer et al., 2008) and this was certainly the case in the current study where all dead trees contained hollows. However, the current study as well as that of Moloney et al. (2002) demonstrated the significant contribution of live trees to the availability of hollows overall. The importance of living HBTs is emphasised given that the persistence of dead trees across the landscape is likely to be less than that of the living cohort of HBTs due to the impacts of fire and wind. Consequently, even though standing dead trees are less abundant in urban forest patches they may persist for longer periods than those in managed forests.

Lophostemon confertus dominated the woody tree communities across the urban forest patches and were also well represented as HBTs. There was a greater prevalence of butt hollows in *L.confertus* than any other species. Unlike other eucalypt species where bark thickness ranges from 13-35 mm, *L.confertus* with a maximum bark thickness of 3 mm has reduced regenerative potential, making them susceptible to fire damage (Burrows, 2002). This may facilitate the formation of butt hollows in larger *L.confertus* that are repeatedly burnt.

Tree-form was related to the presence of hollows and highlighted the importance of living trees, as forms 1 and 2 had significantly more hollows than forms 3-8. These results are

consistent with previous studies establishing that hollows are most likely to occur in large trees with moderately senescent crowns (Smith and Lindenmayer, 1988; Lindenmayer et al., 1991; Whitford, 2002). Fauna have also been observed more frequently (45%) leaving hollows found in the crowns of trees of forms 1 and 2 than other forms (Smith and Lindenmayer, 1988; Lindenmayer et al., 1991).

Diameter at breast height for HBTs ranged from 10-160 cm across all sites, but the mean dbh for HBTs was significantly larger than that for the entire wooded community within the forest patches. These findings support previous studies confirming that larger, older trees have a greater likelihood of containing hollows (Wormington et al., 2003; Koch et al., 2008), as well as having a greater number of hollows per tree (Whitford, 2002; Wormington et al., 2003). The size class distribution of HBTs was also different to that of non-hollow trees with the latter appearing to be of a relatively even age (78% of all trees found within the 20-40 cm dbh range). This potentially reflects the intense logging history of the region where historical forestry practices removed most large mature trees from forest patches and entire fragments were cleared for housing within urban areas (Kowald, 1996; Powell, 1998).

Hollows with a threshold diameter ≤ 20 cm were in the majority within urban habitat remnants on the Gold Coast. While confirming that small hollows are typically more common than larger hollows (Gibbons et al., 2000; Wormington et al., 2003) the relative paucity of large hollows may be limiting for large-bodied vertebrates. Furthermore, larger hollows inevitable take longer to form (Wormington and Lamb, 1999; Gibbons et al., 2000). Hollow-dependent fauna restricted to urban forest remnants may therefore reflect the current urban extinction debt (Tilman et al., 1994), as the natural replacement of hollow trees is beyond the time frames that would ensure persistence of these fauna in urban remnants.

This shortage could be addressed by the installation of species appropriate artificial nest-boxes (Goldingay et al., 2007). Nest-boxes are used by fauna in a range of environments and occupancy may be affected by a number of factors including the density of natural tree hollows, the age of the forest habitat, the time since establishment of the boxes, the size of wildlife populations, the presence of competing species, and nest-box design (Durant et al., 2009; Goldingay and Stevens, 2009; Lindenmayer et al., 2009; Ball et al., 2011). While nest-boxes are known to be used in urban remnants (Harper et al., 2005b; Davis et al., 2013) these authors also called for a greater understanding of HBT resource availability, the presence of species that require tree-hollows, as well as species-specific hollow requirements prior to introducing nest-boxes into any environment. The current study therefore provides essential data about the HBT resource to inform future management practices aimed at providing a wider range of hollow options for urban wildlife.

Conclusion

The City of Gold Coast has a unique set of geographical, historical and economic factors that drive urban planning and natural area management. Consequently, there will be considerable variability in the assessment of habitat resources within such urban landscapes compared to other regions within Australia (e.g. Harper et al., 2005; Le Roux et al., 2014b) and elsewhere (McKinney, 2008). Our study has shown that many forest patches along an urbanisation gradient are relatively intact when considering their ability to provide habitat features. As such they provide an important functional role within these urban landscapes.

However, urban forest remnants are far from secure (van der Ree and McCarthy, 2005). Ongoing urban development that contributes to fragmentation and isolation, continues to exacerbate historical logging effects. Forest patches are therefore relatively young and may

lack the old trees required to provide large hollows. To ensure the future provision of these habitat features to support hollow-dependent fauna we recommend a number of management interventions.

Firstly, efforts should be made to halt the erosion of these habitat remnants, as they may provide important refuge sites for urban fauna by providing a more stable resource base. This is supported by recent research modelling the persistence of HBTs in heterogeneous urban landscapes (Le Roux et al., 2014b). The conservation of HBTs alone, however, will only answer the immediate need for this resource. It is therefore imperative to plan for future needs by conserving forest regrowth as recruitment trees for the long-term sustainability of HBTs. Secondly, a greater understanding of the use of these available hollows is required. In the context of the urban landscape it is also important to determine the relative use of natural hollows over other habitat niches (e.g. urban structures). For example, this may affect the success of any efforts to increase the availability of hollows through the placement of best-boxes. Finally, Treby et al. (2014) have shown that there is little recognition of the significance of HBTs at the local level resulting in the poor transfer of policy into implementation. There is consequently an urgent need to put measures in place within local urban planning provisions that recognise the value of these habitat features and their important functional role. Without these, we risk the ongoing decline of urban biodiversity.

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Donna Treby Figure 1: Map showing the 45 research sites studies within the Gold Coast, Brisbane, Logan and Redland Shires, south-east Queensland, Australia.

Donna Treby Figure 2: Tree form characteristics and senescence adapted from Whitford (2002). **S1.** Mature living tree, **S2.** Mature, living tree with a dead or broken top, **S3.** Living tree with most branches still intact, **S4.** Living tree with the top 25% broken off, **S5.** Living tree with the top 25-50% broken away, **S6.** Living tree with the top 50-75% broken away, **S7.** A solid dead tree with 75% of the top broken away, **S8.** Dead.

Donna Treby Figure 3: Tree hollow types (Lindenmayer et al., 2000).

Donna Treby Figure 4: Number of trees surveyed in each dbh size category.

Donna Treby Figure 5: The number of hollows in each hollow size category.

Donna Treby Figure 6: The relationship between hollow type and size (mean $\pm$ S.E). The number of hollows for each hollow type are; Butt 269, Branch-end 735, Bayonet 914, Fissure 48, Trunk-main 98, Trunk-top 54, Branch-main 25.

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610 Figure 1: James Guy Castley

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Figure 3: Lindenmayer et al., 2000

Figure 4: Donna Treby

Figure 5: Donna Treby

Figure 6: Donna Treby

Table 1: Summary of tree and tree-hollow variables quantified at each plot.

Variable	Description of variable
Tree variables	
Tree genera	<i>Angophora, Corymbia, Eucalyptus, Lophostemon, Syncarpia.</i>
dbh ≥ 10 cm	Diameter at breast height for all trees.
Tree form & senescence	Figure 2.
Number of hollows	Number of hollows observable in a tree from the ground.

Hollow variables

Hollow height (m)	Height of hollow in tree.
Hollow type	Figure 3.
Hollow entrance size (cm)	Approximation of entrance width.
Hollow density	Equation1

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Table 2: A summary of species dominance and hollow-bearing tree (HBT) prevalence recorded within eucalypt communities from 45 forest patches.

Tree species	N sites	N trees	N HBTs	Proportion
<i>Lophostemon confertus</i>	35	968	142	0.15
<i>Corymbia intermedia</i>	44	778	64	0.08
<i>Eucalyptus carnea</i>	30	563	76	0.13
<i>E.crebra</i>	33	527	91	0.17
<i>E.propinqua</i>	33	477	80	0.17
<i>C.citriodora.v</i>	23	417	54	0.13
<i>C.gummifera</i>	16	347	35	0.10
<i>E.tindaliae</i>	15	299	36	0.12
<i>E.microcorys</i>	26	293	71	0.24
<i>L.suaveolens</i>	14	277	20	0.07
<i>E.pilularis</i>	17	216	29	0.13
<i>E.siderophloia</i>	15	162	15	0.09
<i>E.racemosa</i>	6	114	36	0.32
<i>Dead</i>	39	124	124	1.00
<i>E.resinifera</i>	10	96	1	0.01
<i>Angophora woodsiana</i>	3	61	0	-
<i>E.grandis</i>	5	40	2	0.05
<i>E.longirostrata</i>	4	40	0	-
<i>C.trachyphloia</i>	3	36	2	0.06
<i>E.acmenoides</i>	6	30	8	0.27
<i>E.eugenoides</i>	5	31	4	0.13

<i>E.tereticornis</i>	8	25	9	0.36
<i>E.baileyana</i>	1	26	4	0.15
<i>E.major</i>	7	20	9	0.45
<i>E.robusta</i>	1	24	0	-
<i>E.saligna</i>	2	14	1	0.07
<i>C.henryi</i>	3	11	1	0.09
<i>E.moluccana</i>	4	10	0	-
<i>E.biturbinata</i>	4	10	1	0.10
<i>C.tessellaris</i>	1	4	0	-
<i>E.fibrosa</i>	1	3	0	-
<i>E.seeana</i>	2	3	0	-
<i>E.dura</i>	1	1	0	-
<i>Syncarpia glomulifera</i>	1	1	0	-
Totals: 34 species		6048	916	

Table 3: A description of hollow-bearing tree (HBT) attributes found across all sites sorted by area. Site number and locations are reflected in Figure 1.

		No:		N trees	Dominant	Dominant HBT	Mean dbh of	N HBT	N
Site No:	Gradient	plots	Area (ha)	tagged	species	Species	HBTs	species	HBTs
*1	C/P	5	1668.76	387	<i>E.carnea</i>	<i>E.crebra</i>	42.6	12	86
*2	C/P	7	824.33	400	<i>E.tindaliae</i>	<i>E.tindaliae</i>	38.1	13	53
*3	C/U	6	725.07	301	<i>L.confertus</i>	<i>L.confertus</i>	51.2	8	57
*4	C/P	6	434.24	236	<i>E.propinqua</i>	<i>E.propinqua</i>	52.5	11	40
*5	C/U	5	432.06	235	<i>E.tindaliae</i>	Dead	41.6	8	33
6	P	2	565.87	152	<i>C.intermedia</i>	<i>C. intermedia</i>	21.5	1	1
7	P	3	266.07	135	<i>C.citriodora.v</i>	<i>C.citriodora.v</i>	41.0	6	28
8	P	3	185.31	164	<i>C.gummifera</i>	<i>E.propinqua</i>	60.4	11	33
9	P	1	176.87	68	<i>E.racemosa</i>	<i>E.racemosa</i>	48.9	4	14
10	P	2	91.08	117	<i>L.confertus</i>	<i>L.confertus</i>	51.0	7	25
11	P	1	33.67	41	<i>E.microcorys</i>	<i>E.propinqua</i>	39.6	7	15
12	P	1	26.61	85	<i>L.suaveolens</i>	<i>L.suaveolens</i>	42.8	3	8
13	P	1	26.53	83	<i>L.confertus</i>	<i>L.confertus</i>	28.1	8	26
14	P	2	13.10	188	<i>C.intermedia</i>	<i>E.propinqua</i>	52.4	8	34
15	P	2	13.06	210	<i>L.confertus</i>	<i>L.confertus</i>	46.3	8	31
16	P	1	9.15	53	<i>C.intermedia</i>	<i>C.citriodora.v</i>	43.6	4	8
17	P	1	7.66	73	<i>C.intermedia</i>	<i>E.crebra</i>	43.7	4	5
18	P	1	2.97	72	<i>E.pilularis</i>	<i>E.microcorys</i>	45.5	3	4
19	P	1	2.11	85	<i>C.intermedia</i>	<i>E.propinqua</i>	38.5	6	12
20	R	2	157.55	92	<i>C.gummifera</i>	<i>E.microcorys</i>	43.1	6	16
21	R	3	24.98	318	<i>L.confertus</i>	<i>E.microcorys</i>	64.2	6	29
22	R	1	14.29	71	<i>C.intermedia</i>	<i>C.intermedia</i>	58.9	4	11
23	R	1	10.82	72	<i>E.pilularis</i>	<i>E.pilularis</i>	69.6	4	8
24	R	1	5.99	47	<i>L.confertus</i>	<i>E.propinqua</i>	58.4	5	11
25	R	2	383.67	71	<i>E.microcorys</i>	<i>L.confertus</i>	51.3	5	24
26	R	2	193.24	144	<i>C.citriodora.v</i>	Dead	40.2	4	5
27	R	2	53.32	203	<i>E.crebra</i>	<i>E.carnea</i>	47.4	7	26
28	R	1	50.87	111	<i>L.suaveolens</i>	<i>E.siderophloia</i>	51.0	2	2
29	R	3	44.85	178	<i>L.confertus</i>	<i>E.carnea</i>	52.5	8	22

30	R	1	14.24	93	<i>L.confertus</i>	<i>E.microcorys</i>	47.8	6	13
31	R	1	5.10	51	<i>E.microcorys</i>	<i>E.microcorys</i>	77.3	5	16
32	R	1	4.04	69	<i>L.suaveolens</i>	<i>C.citriodora.v</i>	36.2	6	10
33	U	2	38.94	134	<i>A.woodsiana</i>	<i>E.pilularis</i>	60.0	2	2
34	U	1	27.32	86	<i>L.confertus</i>	<i>L.confertus</i>	57.3	4	26
35	U	2	17.89	159	<i>L.confertus</i>	<i>E.crebra</i>	52.8	5	34
36	U	2	16.89	205	<i>L.confertus</i>	<i>E.pilularis</i>	60.1	8	33
37	U	2	16.87	140	<i>C.citriodora.v</i>	<i>E.carnea</i>	38.5	6	22
38	U	3	16.45	198	<i>C.intermedia</i>	<i>E.propinqua</i>	50.3	8	23
39	U	1	14.38	67	<i>E.pilularis</i>	<i>E.pilularis</i>	78.4	4	9
40	U	1	9.53	52	<i>E.microcorys</i>	<i>E.microcorys</i>	54.3	6	14
41	U	1	7.52	50	<i>L.confertus</i>	<i>L.confertus</i>	41.8	3	5
42	U	1	5.89	82	<i>E.crebra</i>	<i>L.confertus</i>	20.2	2	6
43	U	1	4.41	126	<i>E.carnea</i>	<i>L.confertus</i>	34.9	4	5
44	U	1	4.15	66	<i>E.racemosa</i>	<i>E.racemosa</i>	67	3	25
45	U	1	3.23	78	<i>C.intermedia</i>	<i>E.pilularis</i>	39.3	5	6
							Mean	Mean	
Total: 45		91	6650.9	6048	14	13	48.5 ±1.84 cm	5.7 ±0.4	916

†highlighted areas represent the congruency between the dominant tree species found on a site and the dominant HBT species found on a site.

* control sites.

Table 4: Summary of trees species and hollow type with the total number of hollows found at all sites.

Tree species	Bayonet	Branch end	Branch main	Butt	Trunk main	Trunk top	Fissure	Total
<i>Dead</i>	113	135	1	27	24	34	12	346
<i>L.confertus</i>	51	38	0	122	12	4	9	236
<i>E.propinqua</i>	104	93	4	6	8	2	0	217
<i>E.crebra</i>	106	74	0	4	3	0	4	191
<i>E.microcorys</i>	94	57	5	20	6	1	5	188
<i>E.carnea</i>	80	51	1	25	9	4	4	174
<i>C.citriodora.v</i>	81	49	3	10	7	5	2	157
<i>E.racemosa</i>	38	42	6	8	8	2	1	105
<i>C.intermedia</i>	38	34	0	7	4	0	1	84
<i>C.gummifera</i>	49	21	0	3	5	2	0	80
<i>E.tindaliae</i>	40	26	1	8	3	1	1	80
<i>E.pilularis</i>	20	36	1	11	3	0	2	73
<i>E.tereticornis</i>	17	36	1	3	2	0	1	60
<i>L.suaveolens</i>	23	12	0	9	2	0	1	47
<i>E.siderophloia</i>	24	11	0	0	2	0	0	37
<i>E.major</i>	16	6	0	1	0	0	0	23
<i>E.grandis</i>	3	15	0	0	0	1	0	19
<i>E.acmenoides</i>	3	5	2	3	1	1	0	15
<i>E.resinifera</i>	8	3	0	2	0	0	0	13
<i>E.baileyana</i>	2	2	0	0	1	0	0	5
<i>C.trachyphloia</i>	1	4	0	0	0	0	0	5
<i>E.eugenioides</i>	2	0	0	0	0	0	0	2
<i>E.moluccana</i>	1	0	0	0	0	0	0	1
<i>C.henryi</i>	0	1	0	0	0	0	0	1
Total	914	751	25	269	100	57	43	2159

Highlights

- 19 1. Hollow-bearing tree density was high (mean=37.5HBTs/ha) along the entire urban
20 gradient.
- 21 2. Standing dead trees contained the greatest number of hollows but were not as prevalent as
22 living trees with hollows.
- 23 3. Large hollows are relatively uncommon within urban forest patches.
- 24 4. Forest remnants provide key habitat features and therefore maintain functional urban
25 ecosystems.

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