1 Can ground-based assessments of forest biodiversity reflect the biological

2 condition of canopy assemblages?

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21 **Abstract**

22 Biological assessments of forest systems often involve a single ground-

23 invertebrate sampling method that may ignore the biological component of the

24 non-sampled canopy. Pitfall trapping for ground-active arthropods is a widely

25 implemented technique for biological assessment in forested and open habitats.

26 Although much evidence highlights the biases of pitfall trapping, this evidence

27 typically comes from open-habitat crop and grassland systems. In forest systems

28 where much of the biodiversity is found within the above-ground structure,

29 management recommendations based solely on ground sampling may not

30 represent the diversity within the three dimensional forest habitat. We provide

31 evidence from combined ground and canopy sampling of three major forest

32 types within the study region. We use canopy insecticide fogging to compare

33 with more traditional ground-based pitfall trapping, and use spiders as a

34 comparative species-rich biota that is able to colonise most terrestrial habitats

35 and is strongly affected by changes in environmental condition.

36 We identified 3933 spiders from 109 species from the 18 forest patches sampled.

37 Both types of sampling defined differences in community composition between

38 forest types in a similar manner; hence, either method could be used to evaluate

39 differences or test management regimes in well-replicated experiments of forest

40 type. However, the association in community composition between ground and

41 canopy assemblages at the individual site-based level was weak; we found low

42 correlation between the two data sets indicating that surrogacy between

43 methods was not supported at this level. Furthermore, disparities in spider

44 habitat association, body size, hunting guild and vertical stratification of spider

45 families indicates that where detailed species and family-based information is

46   required, or if inventorying is necessary, then multiple targeted surveys are

47   essential.

48

49   Keywords: canopy fogging, arachnid, forest management, arthropod, pitfall,

50   spruce plantation.

51

1. Introduction

Biodiversity must be sampled in a way that fits research questions but also meets time and financial budgets. Often these constraints lead to the use of a single survey procedure to derive data with which to draw conclusions that inform policy and management. This leaves questions regarding the consistency of those conclusions if an alternative sampling strategy had been chosen. In complex systems, such as forested landscapes, the three-dimensional structure poses problems for capturing representative samples across vegetation layers (Pinzon et al. 2011). The importance of forest systems (Ozanne et al. 2003), coupled with the potential of sample bias, means there is a growing need to validate sampling strategies to strengthen management recommendations based on these single survey practices.

Arthropod diversity is frequently used to assess biological condition in applied forest research (Spence et al. 1996, Berndt et al. 2008, Pedley et al. 2014) and more fundamental aspects of ecology, including fragmentation and disturbance (Vasconcelos et al. 2006, Pedley and Dolman 2014). New DNA barcoding techniques (Yang et al. 2014), which negate the often laborious taxonomy associated with arthropod sampling, are enabling quicker processing times that may proliferate the use of arthropod monitoring (Ji et al. 2013). However, conventional taxonomic and many contemporary DNA barcoding techniques rely on traditional invertebrate collection methods. One of the most commonly employed sampling techniques for epigaeic arthropods is pitfall trapping. Pitfall trapping provides a passive means of surveying that, once established, can

76   continuously trap active species with only brief visits needed to service traps.

77   Although pitfall trapping has a long history in ecology, its ability to provide non-

78   bias sampling of habitat has been brought into question (Topping and

79   Sunderland 1992, Lang 2000). Pitfalls by their nature target active ground-

80   dwelling species, and can underrepresent less mobile, small-bodied species and

81   species typical of higher strata (Greenslade 1964, Lang 2000, Standen 2000).

82   Furthermore, pitfall catches are a representation of animal density, conditional

83   on animal activity; if activity is disproportionally affected by vegetation

84   structure, shading or animal interactions between sites, then catches may not be

85   comparable (Greenslade 1964, Melbourne 1999). Where environmental

86   conditions are similar, comparisons across sites are suitable as long as pitfall

87   trap data are used as an index of the density based on activity and not a species

88   inventory of the sampled habitat (Luff and Eyre 1988, Oxbrough et al. 2006).

89

90   Much of the available methodological literature concerning pitfall trap bias

91   comes from crop and grassland studies (e.g. Topping and Sunderland 1992,

92   Standen 2000). However, extensive arthropod monitoring of closed-canopy

93   forests has been conducted with ground-based methods (e.g. Docherty and

94   Leather 1997, Oxbrough et al. 2005, Berndt et al. 2008). Many studies of this

95   nature make comparisons between the arthropod biodiversity of different forest

96   types with inherently different ground, understory and canopy structures (Fuller

97   et al. 2008, Barsoum et al. 2014). Although such studies do not imply that pitfall

98   trapping will reveal the biodiversity related to the entire three-dimensional

99   structure of the forest, there are few studies that can elucidate the non-sampled

100   aboveground component of forest biodiversity in a similar manner to the

101   methodological papers concerning crop and grasslands (but see Pinzon et al.

102   2011). This problem of the non-sampled biodiversity is perhaps more significant

103   within forest systems as forest canopies contain a large proportion of the total

104   arthropod diversity on Earth (Erwin 1982, Lowman and Wittman 1996).

105

106   While canopy sampling is considerably more challenging than many ground

107   sampling methods due to the difficulties in accessing tree canopies, ground-

108   based insecticide fogging can negate these access problems. Insecticide fogging

109   of canopy-dwelling species has proven a reliable survey method but has received

110   less consideration in temperate and boreal zones than in tropical regions.

111   Canopy fogging has proven an effective way to sample temperate canopy

112   invertebrates and to measure biodiversity patterns within single species, across

113   temporal dynamics and between forest types (Southwood et al. 2005, Hsieh and

114   Linsenmair 2012, Pedley et al. 2014). However, fogging is limited by weather

115   conditions, with at least several hours of dry, still weather required for

116   successful sampling. This method may also overlook some species such as aphids

117   or other phloem feeders (Stork and Hammond 1997), or those within certain life

118   stages, such as within cocoons, retreats or burrows and those attached by silken

119   threads. While these sampling biases will affect inventorying canopy

120   invertebrates in much the same way as pitfall trap biases do for ground-based

121   invertebrates, it is likely that standardised canopy fogging will allow for

122   comparisons to be made across sampled forest sites.

123

124   Among the arthropod groups frequently investigated in ecological surveys,

125   spiders provide an effective means of habitat assessment as they are greatly

126   affected by changes in habitat structure (Duffey 1968, Robinson 1981) and

127   respond quickly to brief or sudden changes in environmental conditions, such as

128   variations in prey density, pesticides, or pollution (Marc et al. 1999). Spiders are

129   a species rich group and, being one of the top macro-invertebrate predators,

130   have strong influences in food webs (Wise 1993, Schmitz et al. 2000). Differences

131   in spider community assemblages within forest types have often been attributed

132   to differences in habitat heterogeneity (Pinzon et al. 2011, Pedley et al. 2014).

133   The assemblage composition of the forest-floor is influenced by light availability,

134   volume and decay stage of debris, moisture and temperature (Ziesche and Roth

135   2008); while canopy leaf/needle density and branch architecture has been

136   shown to influence community composition above the ground layer (Gunnarsson

137   1992, Halaj et al. 2000). Although some understanding of the factors influencing

138   community composition in these habitats exists, we do not yet know if common

139   sampling techniques differentially interpret community dissimilarities between

140   forest types.

141

142   In the current study, we selected three distinctive forest types that were likely to

143   vary in spider composition, semi-natural ash (*Fraxinus excelsior*) forests, semi-

144   natural oak (*Quercus petraea*) forests and Sitka spruce (*Picea sitchensis*)

145   plantations. We did not attempt to directly compare species richness or

146   abundance between canopy and ground trapping, as sampling effort is not

147   consistent between the two methods. Rather, we examined whether there is

148   correspondence between the two methods for defining differences in

149   assemblage structure between the three forest types. For each of the following

150   hypotheses we looked for idiosyncratic and corresponding changes in

151   biodiversity structure across forest types for the ground and canopy sampling

152   techniques. 1) Assemblages sampled in the canopy and the ground differ

153   similarly between the forest stands and forest types. 2) Patterns of hunting

154   guilds (active and web spinners), habitat specialism (woodland and generalist),

155   and body size will be inconsistent across forest types for ground and canopy

156   sampled assemblages. 3) Spider families will show vertical stratification between

157   ground and canopy sampling. Finally, we discuss whether there is possible

158   surrogacy between ground and canopy methods. This is one of the first studies to

159   compare and interpret forest biodiversity obtained from canopy and ground

160   trapped invertebrate assemblages.

2. Methods

Three closed-canopy forests types were sampled across Ireland (Appendix A); six ash (*Fraxinus excelsior*) dominated semi-natural woodlands , six oak (*Quercus petraea*) dominated semi-natural woodlands and six second-rotation Sitka spruce (*Picea sitchensis*) plantations (hereafter referred to as ash forest, oak forest and spruce plantation, respectively). All stands were a minimum of 6 ha in size and 100 m in width. Sitka spruce plantations were selected as they are the dominant species in the Irish forest estate, comprising approximately 60% of the forest cover and are a non-native species (Forest Service 2007). Ash and oak forests were selected as they are the most common native tree species in Irish semi-natural forests, comprising 22% and 18%, respectively (Higgins et al. 2004), and were expected to have contrasting biodiversity to spruce plantations. The semi-natural forest types considered in this study comprised a mix of tree species, i.e. oak-dominated forests included oak, birch and holly, while ash-dominated forests included ash, oak and hazel. Semi-natural ash and oak forests were at least 150 years old, whereas sampled spruce plantations ranged from mid rotation 20-30 year old closed-canopy stands to 60-year-old commercially mature stands.

2.1 Canopy sampling

Canopy fogging was conducted once at each of the 18 study sites. In each sampled forest stand a fogging plot was established in a representative area of the site in terms of stand structure and vegetation cover. A target tree was selected at the centre of each fogging plot that corresponded to the forest type being sampled. The fogging plot consists of 16 plastic sheets (1.5m^2), with a

combined area of 24m^2, arranged around the central tree on the eight cardinal
and ordinal compass bearings. Plastic sheets were suspended 1m above the
ground to reduce the risk of contamination by ground-active species not
sampled by fogging (Stork and Hammond 1997). Sampling sheets were
separated by 0.5m from each other and all trees within the fogging plot. The
fogging plot was at least 50m from the forest perimeter to reduce possible edge
effects.

Fogging was carried out between April and August in 2008 and 2009. A petrol-
driven fogging machine (SwingFog SN50-PE, SwingTec Ltd, Germany) was used
with a natural pyrethroid (Pybuthrin 33). Pyrethroid insecticide was chosen as it
is non-persistent in the environment, with no phytotoxic effects and the levels
used by this method are not harmful to mammals (Straw et al. 1996). Each
canopy was fogged until fully covered in insecticide (typically 6-9 minutes
duration). Fogging was only carried out in dry, calm conditions (wind-speeds of
less than 8 km h^{-1}) and after a dry, calm night to minimise fog dispersion. At each
site, sample sheets remained in place for three hours after fogging (Stork and
Hammond 1997), after which the captured invertebrate material was pooled and
stored in 70% alcohol.

2.2 Pitfall sampling

Pitfall trapping was conducted for 63 days at each of the 18 study sites. At each
site, pitfall traps were used to collect ground-dwelling spiders from three
sampling plots. Each sampling plot was 50m apart and 50m away from the forest
edge and comprised a transect of five pitfall traps spaced 2m apart. Pitfall traps

211   consisted of a plastic cup 7cm in diameter and 9cm in depth. Traps were filled

212   with ethylene glycol (1cm deep) to act as a killing and preserving agent. All traps

213   were set in mid-May 2007 and left *in situ* for nine weeks with traps serviced

214   every three weeks. There was considerable animal disturbance (> 80% trap loss)

215   at two of the spruce study sites sampled during 2007 and so these sites were re-

216   sampled during the same period in 2008 with previous material being discarded.

217   No other trap disturbance was recorded during the study. Catches from each site

218   were pooled across the nine weeks for analyses giving a total of 945 trap days

219   (63 days x 5 traps x 3 plots) per site.

220

221   **2.3 Analysis**

222   Abundance measures for analysis comprise the numbers of individuals per

223   canopy plot and numbers of individuals per pitfall plot (pooled across traps and

224   sampling periods for pitfall traps). All analysis was carried out in the statistical

225   software R v3.1.2 (R Development Core Team 2012). To visualise the difference

226   in richness and abundance of spiders recorded from the different sampling

227   methods, we calculated individual-based rarefaction curves using the rarecurve

228   function in the vegan package (Oksanen et al. 2010).

229

230   Indicator species analysis was conducted separately for the two sampling

231   methods to determine species affinity to forest types. We used the function

232   multipatt in the package indicspecies (De Cáceres et al. 2010) to calculate species

233   indicator values (Dufrêne and Legendre 1997), and permutation (999) to test the

234   significance.

235

11

236   To visualise the community composition among forest types for each sampling

237   method, we used non-Metric Multidimensional Scaling (NMDS), performed on a

238   matrix of Bray-Curtis dissimilarities of abundance data (square root transformed

239   and Wisconsin double standardization) using the vegan package. Differences in

240   community composition between forest types were tested using the package

241   mvabund (Wang et al. 2012), which allows hypothesis testing by multivariate

242   implementation of generalised linear models. This method does not confound

243   location with dispersion effects (a change in the mean-variance relationship),

244   which can lead to misleading results and inflation of type-1 and 2 errors (Warton

245   et al. 2012). Using likelihood-ratio-tests (LR) in the summary.manyglm function

246   we tested for significant differences in assemblage composition of spruce and

247   semi-natural ash and oak forests.

248

249   We use Procrustes rotation analysis on NMDS scores to explore the degree of

250   congruence between the different biotic datasets obtained by pitfall trapping and

251   fogging (Peres-Neto and Jackson 2001). We implemented the protest function in

252   the vegan package to test the best fit of two ordinations against a relationship

253   occurring by chance (Peres-Neto and Jackson 2001); larger correlation

254   coefficients indicate a better concordance between two datasets (perfect

255   concordance when correlation coefficient=1).

256

257   Species richness and abundance of woodland associated and generalist species

258   were compared among forest types separately for ground and canopy sampling

259   using generalised linear models (GLMs). The appropriate error term for each

260   analysis was selected by patterns in residuals and by examining model

261   dispersion. Differences among forest type means were examined by Tukey

262   pairwise comparisons using the glht function in the multcomp package (Hothorn

263   et al. 2008). Spatial autocorrelation of GLM residuals was examined by Moran's I

264   in the ape package v.3.0-6 (Paradis et al. 2004). In all instances, Moran's I was

265   not significant ($P > 0.05$).

266

267   For each sampling site we calculated the abundance-weighted mean values for

268   spider body size. This metric simply multiples spider body-size by the sampled

269   abundance of each species recorded per site and calculates a single community-

270   weighted mean (CWM) per site. Body size for each species was obtained from

271   Roberts (1987, 1996). We tested the average CWM body-size of spiders sampled

272   by pitfall trapping against those sampled by fogging using a Man-Whitney U test.

273   Within each sampling method, differences in CWM body-size between forest

274   types were tested with GLMs as above.

275

3. Results

We identified 3933 spiders of 109 species from the 18 forest patches sampled. Pitfall trapping, where each of the 15 traps per site was active for 63 days as opposed to a single discrete trapping event per site for fogging, captured a greater overall abundance and richness of spiders (Fig. 1). Pitfall trapping captured 3205 spiders from 87 species whereas fogging captured 728 spiders from 36 species. Analysis of variance showed that species richness of the three forest types was significantly different for pitfall-trapped assemblages ($F_{2,15}$=5.141, P=0.020; ash mean (±SD) 23.8±4.6, oak mean 25.7±5.7, spruce mean 17.7±2.6), with post hoc tests indicating that only oak and spruce were significantly different (Turkey P=0.020). No differences in species richness were found between forest types for canopy-fogged assemblages (Kruskal-Wallis χ^2=0.467, P>0.05; ash mean (±SD) 6.8±2.2, oak mean 6.7±0.5, spruce mean 7.0±2.6). For details of species identification and classification of hunting and habitat guilds see Appendix B.

Fourteen species were common to both trapping methods, 73 species were unique to pitfall trapping (including 22 woodland species) and 22 to fogging (including five woodland species). Of the 87 species in pitfall traps, 16 (18%) species were unique to ash forest, 14 (16%) unique to oak and five (6%) unique to spruce. A larger proportion of species were unique to forest types in the fogged samples, from the 36 species captured, nine (25%) were unique to ash forests, seven (19%) were unique to oak forest and nine (25%) were unique to spruce plantations. Twenty five (29%) species were common to all forest types

300   sampled by pitfall trapping whereas six species (16%) were common to all

301   fogged forest types.

302

303   The woodland-associated Linyphiid, *Lepthyphantes zimmermanni* was the most

304   abundant species recorded in all forest types by pitfall trapping (ash n=331, oak

305   n=293, spruce n=140). For canopy assemblages, ash forests were dominated by

306   the generalist species *Tetragnatha montana* (n=66), oak forests by the woodland

307   species *Neriene peltata* (n=64) and spruce by the woodland species *Pelecopsis*

308   *nemoralis* (n=184).

309

310   Indicator species analysis identified associations for all forest types, although

311   canopy sampled indictors were only identified from ash and spruce (Table 1). All

312   species identified as indicators were web hunters. The highest indicator values

313   for pitfall-trapped species were for habitat generalist whereas those sampled by

314   fogging were both generalist (ash) and woodland associated (spruce).

315

316   Significant differences in community composition were found between forest

317   types using both pitfall trapping and fogging methods (Deviance = 360.6,

318   P=0.007; Deviance=137.4, P=0.004 respectively); however, assemblages sampled

319   with pitfall trapping showed greater separation between forest types (Fig. 2).

320   Compared with spruce, semi-natural forests had significantly different species

321   compositions for both pitfall trapped and fogged assemblages (P<0.001).

322   Although fogged ash and oak forests showed some overlap in NMDS space,

323   assemblages in the two semi-natural forest types were significantly different (LR

324   value=45.61, P=0.002).

325

Procrustes tests performed on the NMDS ordinations indicated significant

concordance between pitfall trapped and fogged assemblages ($m^2=0.77$,

correlation coefficient = 0.48, P=0.026); however, this was not sufficiently strong

(m^2=<50, correlation coefficient >0.7) to regard robust surrogacy between

sampling methods (Heino 2010).

331

Abundance and richness of woodland associated species were similar in all three

forest types for pitfall sampled assemblages (Fig. 3a and e, Appendix C). Fogged

spruce assemblages contained significantly more woodland species than semi-

natural forests (Fig. 3f); however, only ash forests had significantly lower

woodland abundance than spruce (Fig. 3b). The fogged spruce assemblage

contained significantly fewer generalist species than ash forests (Fig. 3d) and

showed a general trend of reduced generalist abundance. However, it should be

noted that richness measures obtained from canopy fogging may be less reliable

due to the low abundances caught via this sampling method. Spruce plantations

sampled by pitfall traps contained significantly less generalist richness than ash

and oak forests (Fig. 3g), and generalist species abundance was significantly

higher in oak forests than either spruce or ash forests (Fig. 3c).

344

Very few species of active hunting spider were recorded in either pitfall

assemblages (n=10, mean per sample = 0.8±1.1 SD) or fogged assemblages (n=2,

mean per sample = 0.1±0.3 SD). In addition, no active hunting spiders were

captured in spruce plantations (Fig. 4). Pitfall assemblages in spruce plantations

16

349 were confined to two web-hunting families, Linyphidae and Theridiidae; fogging

350 assemblages also included web hunters from the family Tetragnathidae (Fig. 4).

351

352 CWM body size differed significantly between spiders sampled by pitfall traps

353 and fogging (U=90, P=0.022, pitfall trapping: mean (±SD) 5.1±2.7, range 1.7–

354 13mm; fogging: mean 3.2±2.3, range 8.2–0.5mm). In pitfall-trapped assemblages,

355 spider body-size was smaller in spruce than semi-natural forests, although the

356 only significantly difference in CWM spider body size was between oak and

357 spruce (χ^2 =1.62, P=0.011, ash mean (±SD) 5.6±1.4, oak mean 6.5±3.7, spruce

358 mean 3.1±1.2). No significant differences in body size were found between forest

359 types sampled by fogging (ash mean (±SD) 3.8±3.3, oak mean 2.9±1.5, spruce

360 mean 2.8±2.0).

361

4. Discussion

To explore possible congruency in biological assessment methods, we evaluated ground-based pitfall trapping compared to canopy insecticide fogging of spiders from three closed-canopy forest types. Both pitfall trapping and canopy fogging separated spider assemblages of managed plantation forest from semi-natural forest types. Both methods indicated that the greatest differences in assemblage composition were between ash and spruce plantations. Despite these broadly similar patterns in composition, congruency between sampling methods was not strongly supported, specifically, procrustes rotation produced low correlation scores. This illustrates that at the broadest scale of forest type (oak, ash and spruce) differences in community composition were consistent between sampling methods, but the between site differences were not consistent enough to allow surrogacy in methods. Therefore, unless surveys are designed to look specifically at broad scale patterns in well-replicated studies, forest assessments of spider community assemblages require separate sampling of forest layers. However, where a single sampling method is implemented a clear statement of the bias is essential.

Pitfall trapping recorded greater species richness in ash and oak assemblages than spruce plantations. This is consistent with previous research showing low species richness of ground-dwelling invertebrates within managed coniferous forest sampled by pitfall trapping (Finch 2005, Fuller et al. 2008). In contrast, no differences in species richness were detected between forest types surveyed by fogging. The divergent patterns in ground and canopy richness may provide evidence for stratified biodiversity patterns between forest layers and may

18

387   relate to comparative differences in the habitat heterogeneity of forest stratums.

388   Although it has been shown that branch composition and leaf density influence

389   arthropod composition (Gunnarsson 1992, Halaj et al. 2000), it is possible that

390   differences in habitat heterogeneity within the canopies of the three forest types

391   is not as influential to spider richness as heterogeneity at the ground layer.

392   However, it should also be noted that the disparity in patterns of species

393   richness between sampling methods might also be related to the uneven

394   sampling effort between surveys at different forest layers (Pinzon et al. 2011).

395   For example, canopy sampling may not have been comprehensive enough to

396   detect differences in coarse measures such as species richness. Rarefaction for

397   fogging showed that species richness curves for ash and oak were steeper than

398   spruce plantations, indicating that the sampling in these sites was not as

399   complete. Greater sampling effort, i.e. more trees fogged per forest patch, may

400   detect a larger disparity between semi-natural and plantation forests. However,

401   our relative sampling effort is likely to be reasonable and consistent with other

402   studies (see Zheng et al. 2015, Yanoviak et al. 2003) given the logistical

403   difficulties and the labour intensive nature of this method.

404

405   **4.1 Hunting guilds, habitat specialism and body size representation**

406   Web-hunting Linyphiids dominated the assemblages of both survey methods.

407   Active hunters were represented by very few individuals in semi-natural forest

408   types and there was a total absence of active-hunting species in spruce

409   plantations for both trapping methods (proportion of active hunters: ash 1.8%,

410   oak 0.5%). This is consistent with previous surveys of plantation forest

411   conducted using pitfall trapping in the same region (Oxbrough et al. 2010, Fuller

et al. 2014). Barsoum et al. (2014) compared both Irish and English spider

assemblages and found forest plantations in Ireland to be dominated by web-

hunting Linyphiidae, whereas assemblages in England comprised a mixture of

hunting guilds. In North American broad-leaf forests, Larrivée & Buddle (2009)

found 21 species of hunting spiders from understory and canopies sampled by

beating; while sweep netting by Stratton et al. (1978) found hunting spiders in

the understories of three types of North American coniferous forest. The paucity

of active hunters in the Irish forest fauna may be a result of meteorological

conditions favouring smaller species such as the web-hunting Linyphiidae

(Entling et al. 2010). This combined with the high dispersal potential of

Linyphiidae that are able to balloon as adults over vast distances (Thomas et al.

2003, Bell et al. 2005), may help to explain their dominance in the severely

fragmented Irish forest system where less than 1% of the land cover was

forested at the end of the 19th century (Forest Service 2007, Forest Europe et al.

2011).

It might be expected that older forests would accumulate more forest specialists

and hence old growth and/or semi-natural forests would contain a greater

diversity of specialists than relatively young plantations (Niemelä 1997, Fuller et

al. 2008). However, pitfall trapping indicates no significant difference in

specialist woodland species abundance or richness between forest types.

Moreover, fogging shows more woodland species abundance and richness in

spruce than in natural forest and the majority of indicator species of spruce were

woodland associated whereas most indicators of semi-natural forests were

generalists. Pawson et al. (2008) found that mature exotic plantation forests

437   were able to support native forest beetles and compared to other non-native

438   habitats, such as pasture and clearfell forestry sites, beetle composition in these

439   mature plantations was most similar to native forest. This gradient of landscape

440   suitability was also proposed by Brockerhoff et al. (2008) who suggest that

441   plantation forest could provide useful habitat where pre-plantation areas are

442   non-natural habitat. In the Irish context, plantation forest is not replacing semi-

443   natural forested and it is unlikely the woodland specialists identified in the

444   current study would be found in open habitat in the region (Oxbrough et al.

445   2006, 2007). Therefore, given that semi-natural forest in the Irish landscpe is

446   scarce (1% of total land cover), these areas of plantation maybe important for

447   the canopy fauna, providing essential habitat in an prodominatly open landscape.

448   The disparity between ground and canopy results for woodland associated

449   species highlights the need for greater sampling coverge in forest assessments.

450

451   While we show that pitfall catches provided a larger CWM body-size than

452   fogging, this is not unexpected given the bias of pitfall trapping to select for

453   larger species (Lang 2000), which are typically more active. Interestingly, we

454   were able to detect a significant difference in body size between forest types

455   from ground samples, differences that were not detected from canopy samples.

456   From ground samples, spider body-size tended to be smaller in spruce

457   plantations than oak and ash forests. This is an interesting finding and may

458   result from different moisture and light regimes and would require targeted

459   investigation to confirm the underlying drivers. Previous studies have indicated

460   moisture and climate as potential drivers of body size variation (Wagner et al.

461   2003, Entling et al. 2010). Wagner et al. (2003) demonstrated a reduction in

462   average body size and a change in dominant foraging mode (active to web), and

463   suggested a moisture gradient could be driving stratification in their study of

464   forest spider at various litter depths. At a European scale Entling et al. (2010)

465   looked at size-climate relationships across European spider assemblages and

466   concluded body size decreases from warm/dry to cool/moist climates.

467

468   **4.2 Vertical stratification**

469   While we show stratification of spider families between forest layers, it was not

470   simply that active ground-hunting families such as Lycosidae and Clubionidae

471   dominated the ground catches because all sampling was dominated by small web

472   spinners. Very few studies have attempted to look at both canopy and ground

473   dwelling spider assemblages (but see Docherty and Leather 1997, Pinzon et al.

474   2011, 2013) as vertical stratification of spider guilds in forests is difficult to

475   study given the very different survey strategies required, which can lead to

476   sampling designs that are not comparable. Most studies concerning vertical

477   stratification in temperate and boreal forests have looked at discrete elements,

478   focusing on canopy-understory stratification (Larrivée and Buddle 2009, Aikens

479   and Buddle 2012) or different litter layers (Wagner et al. 2003) where vertical

480   stratification and shifts in family dominance have been reported.

481

482   Although all samples were dominated by Linyphiidae in the current study, pitfall

483   traps did contain five active hunting families in comparison to just two recorded

484   from fogging. Of these families, only Anyphaenidae was recorded from both

485   ground and canopy sampling, indicating strong family stratification of

486   assemblages. In sampled canopies, web-hunters from the families Theridiidae

487   and Tetragnathidae are also abundant. The only non-Linyphiidae species caught

488   in moderate abundance in the pitfall traps was the Theridiidae, *Robertus lividus*,

489   a ground-dwelling habitat generalist. The majority of spiders identified in the

490   indicator analysis in both the canopy and the ground belonged to the same guild:

491   web-hunting species of Linyphiiddae, with *Tetragnatha montana*, a dominant

492   spider in ash canopies, the only exception. Differential representation of spider

493   families between methods in the current study and the large number of species

494   unique to each method (pitfall n=73, fogging n=22) indicates strong

495   stratification. This stratification illustrates the need to incorporate multiple

496   sampling methods across various forest strata if a more complete understanding

497   of the forest fauna is required. This corresponds to the findings from North

498   American spruce where a strong difference in assemblage composition was

499   recorded between forest layers (Pinzon et al. 2011, 2013).

500

501   **Conclusions**

502   Our study set out to examine the correspondence between pitfall trapping and

503   canopy fogging sampling methods for defining differences in spider assemblage

504   structure in several forest types. Our findings show that if fine detailed species

505   and family based information (e.g. habitat association, hunting guild, body size)

506   is required, then separate targeted surveys are needed, as results were not

507   consistent between methods. Furthermore, many species were unique to a single

508   survey methods, so if management priorities are to maintain or increase

509   diversity, then monitoring of both ground and canopy fauna needs to be

510   undertaken. Comparisons using solely species incidence obtained by either

511   survey method should be avoided. This coarse metric reduces data complexity,

512   can be highly susceptible to sampling effort and can be misleading in habitats

513   that contain numerous non-specialist species such as ecotones, disturbed and

514   small-fragmented habitats (Downie et al. 1996, Niemela 1997).

515

516   Associations of community composition between the ground and canopy

517   assemblages were not strong enough to allow surrogacy at the individual site

518   level. Weak correlations between site community compositions imply forest

519   assessments need to include both ground and canopy sampling to provide

520   information on these discrete spider assemblages. In studies where a single

521   sampling method is implemented clear statements of the sampling bias should

522   be incorporated. Although our sampling and analyses indicate discrete ground

523   and canopy assemblages, we were able to define and separate the different forest

524   types using either survey method. Both surveys found the greatest community

525   composition differences between ash and spruce. The fact that both survey

526   methods produced similar outcomes for the broad scale (forest type) community

527   analysis, suggests that either method may be suitable for testing management

528   differences based on spider community assemblages in well-replicated

529   experiments within similar ecosystems. However, it must be emphasized that

530   only the broad variations in composition between forest types are similar, and

531   not the actual compositions as indicated by the weak site-based correlation.

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830 **Table 1.** Indicator species identified from the three sampled forest types

831 showing species habitat specialism, hunting guild and associated test statistics.

Forest type	Sampling method	Family	Species	Habitat association	Hunting guild	Indicator value	P-value
Ash	Pitfall trap	Linyphiidae	*Ceratinella scabrosa*	Generalist	web	0.831	0.013
	Pitfall trap	Linyphiidae	*Lepthyphantes tenebricola*	Woodland	web	0.808	0.039
	Canopy fog	Tetragnathidae	*Tetragnatha montana*	Generalist	web	0.773	0.044
Oak	Pitfall trap	Linyphiidae	*Walckenaeria acuminata*	Generalist	web	0.850	0.012
	Pitfall trap	Linyphiidae	*Microneta viaria*	Generalist	web	0.793	0.039
	Pitfall trap	Linyphiidae	*Walckenaeria dysderoides*	Generalist	web	0.772	0.033
Spruce	Pitfall trap	Linyphiidae	*Centromerus dilutus*	Generalist	web	0.840	0.015
	Pitfall trap	Linyphiidae	*Diplocephalus latifrons*	Woodland	web	0.809	0.042
	Pitfall trap	Linyphiidae	*Monocephalus fuscipes*	Woodland	web	0.727	0.050
	Canopy fog	Linyphiidae	*Pelecopsis nemoralis*	Woodland	web	0.921	0.006
	Canopy fog	Linyphiidae	*Lepthyphantes obscurus*	Woodland	web	0.816	0.030

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Figure descriptions:

Fig. 1. Individual-based rarefaction curves for a) pitfall trapped and b) canopy

fogged spider assemblages from three forest types (ash, oak and spruce).

Fig. 2. Non-Metric Multidimensional Scaling (NMDS) ordination comparing the

spider assemblage composition of three forest types for a) pitfall trapping

assemblages and b) canopy fogged assemblages. Stress scores for each

ordination are 0.14 and 0.15 respectively. Points are sampled sites with lines

connecting to habitat centroids and polygons represent 95% confidence interval

of forest type centroids.

Fig. 3. Mean (± s.e) spider abundance and species richness of woodland and

generalist species per forest type for pitfall trapping and canopy fogging.

Asterisks indicate significant differences from the forest type with the greatest

species richness or abundance in each plot as derived from generalised linear

models (Tukey pairwise comparisons P < 0.05). See Appendix C for model

statistics. Thick central line separates abundance and species richness plots.

Fig. 4. Total species richness per spider family recorded in three forest types

(ash, oak and spruce). Spiders sampled by pitfall trapping are shown in the top

three plots, those sampled by canopy fogging are shown in the bottom three

plots. Families in each plot are split (dotted line) by those families that exhibit a

web-hunting (Web) strategy and those with an active-hunting (Active) strategy.