

The scuttle fly (Diptera: Phoridae) assemblages of a wildfire-affected hemiboreal old-growth forest in Tyresta (Sweden)

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In natural forests, fire is an important disturbance factor and many studies have been carried out concerning its effect on different ecosystems, but no studies have previously been done considering the scuttle flies in hemiboreal forests. Here, we carried out an ecological investigation of the scuttle fly assemblage in a hemiboreal old-growth forest in Tyresta National Park and Nature Reserve (Sweden) from material collected, using Malaise traps, after wildfires in 1997 and 1999. We evaluated abundances of species, dominance structure, species richness (by non-parametric species richness method – *Chao 1*) as well as phenology. The most abundant species of the dominant group (i.e. *Megaselia pleuralis*, *M. nigriceps*, *M. pulicaria*-complex and *M. brevicostalis*) are multivoltine and saprophagous, displaying spring and late summer/autumn activity.

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1. Introduction

Fire represents a prevalent natural factor in the dynamics of boreal and hemiboreal forest ecosystems. The influence of fire on the insect fauna is both short term (direct) and long term through the influence of the subsequent vegetational succession (Holliday 1991). The number and size of wildfires, as well as the “fire-free” periods connected with weather conditions, have important ecological consequences by enhancing local and regional heterogeneity and may be especially important for organisms that colonize recently burnt areas through migration (Danks & Footitt 1989,

Koivula *et al.* 2006, Niklasson & Granström 2000, Southwood 1962, Zackrisson 1977).

In natural forests, fire is an important disturbance factor and many studies have been performed on their effects on different ecosystems (DeBano *et al.* 1998). Wildfire effects on invertebrates have been considered numerous times (Swengel 2001, Wikars & Schimmel 2001, Moretti *et al.* 2002). Wildfires have been shown to increase the species richness of insects, especially among beetles and butterflies (Muona & Rutanen 1994, Toivanen & Kotiaho 2007, Hirowatori *et al.* 2007).

Phoridae (scuttle flies) is one of the most

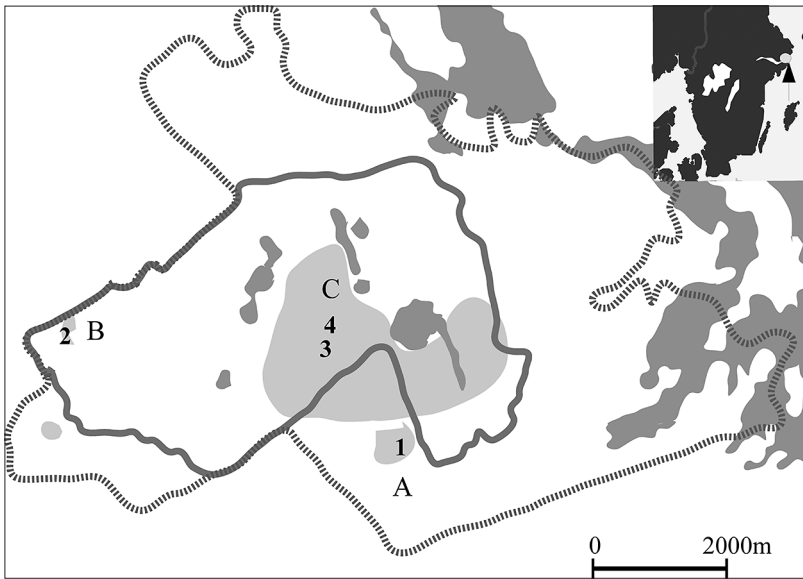


Fig. 1. Location of the Tyresta National Park (continuous line) and the Nature Reserve (dashed line) study area. Dark grey areas indicate lakes and light grey areas wildfire affected forest areas. The black arrow shows the position in Sweden and proximity to Stockholm (white circle). Numbers (1, 2, 3, 4) refer to the position of the traps and letters (A, B, C) to wildfire sites.

abundant and diverse Diptera families in the majority of terrestrial ecosystems (Disney 1994). Scuttle flies also constitute a major part of the pioneer fauna recolonizing habitats after disturbances such as clear cuttings (Durska 2001, 2006, 2009). Prescher *et al.* (2002) studied the scuttle fly assemblages in a chestnut forest in Switzerland after wildfires.

However, no studies have previously been done considering wildfire effects on the scuttle flies in hemiboreal forests (i.e. in the northern part of the temperate zone where the deciduous forest mixes with the taiga).

An extensive arthropod survey that had been running for four years (1999–2002) in a Swedish old-growth forest in the Tyresta National Park and Nature Reserve provided us a unique opportunity to examine the effect of wildfires on the scuttle fly assemblage structure and to improve our knowledge of Phoridae diversity in Sweden. The raw data was problematic since the survey originally was not intended for advanced statistical analyses, i.e., the samples were not exactly from the same locations within the area together with differences in periods and trap days (sampling effort).

As the observed number of species and individuals increase with increased sampling effort the samples may not be comparable i.e., it will likely result in differences in diversity indi-

ces/species richness and may only reflect sampling effort (Scharff *et al.* 2003, Colwell & Coddington 1994). Thus, standardization of the number of individuals and species among sites may improve our understanding of abundance patterns. Non-parametric species richness estimation methods have been used to control for unequal sampling effort (Walther & Martin 2001). The purpose of this paper is to document and consider the abundance, dominance structure and phenology of Phoridae in the Tyresta area near Stockholm.

2. Material and methods

2.1. Material

The material studied was collected in 1999 in Tyresta National Park and Nature Reserve (Tyresta Forest), which is situated 30 kilometers south of Stockholm, Sweden (59°9′–59°13′ N, 18°12′–18°24′ E) (Fig. 1). Scuttle flies were collected using Malaise traps (Marris House Nets™), with collecting heads containing 50% propylene glycol and a few drops of detergent to facilitate submersion of trapped insects. Identifications were done under a dissecting microscope with the material transferred to glycerol. Unidentified *Megaselia* material was set aside as *Mega-*

Table 1. List of species (males) with number of individuals (*n*) given for 1999, together with the number of trap days for each sample. Relative abundance (%) is given for the total sample. Dominant species for the equal sampling effort with dominance $\geq 1\%$ (see Table 2) are shown in bold type. Species with * are new to Sweden, *n* = number of individuals, *T.S* = total sample.

| Sample | 29 | 27 | 9 | 10 | 17 | 19 | 18 | 20 | <i>T.S</i> | <i>T.S</i> |
|---|----------|------------|------------|-----------|-----------|-----------|----------|-----------|------------|-------------|
| Trap days | 24 | 49 | 37 | 44 | 28 | 28 | 42 | 42 | 294 | 294 |
| | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | <i>n</i> | % |
| <i>Anevrina unispinosa</i> (Zetterstedt, 1860) | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| * <i>Beckerina umbrimargo</i> (Becker, 1901) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>Conicera floricola</i> Schmitz, 1938 | – | – | 2 | – | – | – | – | – | 2 | 0.03 |
| <i>Conicera similis</i> (Haliday, 1833) | – | – | 3 | – | – | – | – | – | 3 | 0.05 |
| <i>Conicera tibialis</i> Schmitz, 1925 | – | – | 2 | – | – | – | – | – | 2 | 0.03 |
| <i>Diplonevra nitidula</i> (Meigen, 1830) | – | – | 1 | – | 18 | – | – | 2 | 21 | 0.32 |
| <i>Hypocerides nearcticus</i> (Borgmeier, 1966) | – | – | – | – | 3 | – | – | – | 3 | 0.05 |
| <i>Megaselia abdita</i> Schmitz, 1959 | – | – | 1 | – | – | – | – | – | 1 | 0.02 |
| <i>M. aculeata</i> (Schmitz, 1919) | – | – | – | – | – | – | 1 | – | 1 | 0.02 |
| <i>M. aequalis</i> (Wood, 1909) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. affinis</i> (Wood, 1909) | – | 1 | 5 | 5 | – | – | – | – | 11 | 0.17 |
| <i>M. albicaudata</i> (Wood, 1910) | – | 5 | – | – | – | – | – | – | 5 | 0.08 |
| <i>M. albiclava</i> Schmitz, 1926 | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. albocingulata</i> (Strobl, 1906) | – | 10 | 5 | 3 | 4 | 20 | 1 | 1 | 44 | 0.67 |
| <i>M. annulipes</i> (Schmitz, 1921) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. barbulata</i> (Wood, 1909) | – | – | – | – | – | 1 | – | – | 1 | 0.02 |
| <i>M. basispinata</i> (Lundbeck, 1920) | – | 14 | 19 | 2 | 3 | 9 | – | – | 47 | 0.72 |
| <i>M. berndseni</i> (Schmitz, 1919) | – | 2 | 4 | – | 1 | – | – | – | 7 | 0.11 |
| <i>M. bifida</i> Disney, 1983 | – | 1 | – | – | 1 | 1 | – | – | 3 | 0.05 |
| <i>M. brevicostalis</i> (Wood, 1910) | – | 125 | 185 | 17 | 48 | 65 | – | 8 | 448 | 6.83 |
| <i>M. campestris</i> (Wood, 1908) | – | – | – | 1 | – | – | – | – | 1 | 0.02 |
| <i>M. ciliata</i> (Zetterstedt, 1848) | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. communiformis</i> (Schmitz, 1918) | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. conformis</i> (Wood, 1909) | 2 | 1 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. consetigera</i> (Schmitz, 1925) | – | 13 | – | – | – | – | – | – | 13 | 0.20 |
| <i>M. crassipes</i> (Wood, 1909) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. curvicapilla</i> Schmitz, 1947 | 3 | 27 | 1 | – | – | – | – | – | 31 | 0.47 |
| <i>M. differens</i> Schmitz, 1948 | – | 6 | – | – | – | – | – | – | 6 | 0.09 |
| <i>M. discreta</i> (Wood, 1909) | – | 5 | – | 1 | 4 | 1 | – | – | 11 | 0.17 |
| <i>M. diversa</i> (Wood, 1909) | – | 18 | – | – | 9 | 19 | 7 | 18 | 71 | 1.08 |
| <i>M. drakei</i> Disney, 1984 | – | 6 | – | 2 | – | – | – | – | 8 | 0.12 |
| <i>M. eccoptomera</i> Schmitz, 1927 | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. elongata</i> (Wood, 1914) | – | – | 1 | – | – | – | – | – | 1 | 0.02 |
| <i>M. emarginata</i> (Wood, 1908) | – | 20 | – | 1 | – | – | – | – | 21 | 0.32 |
| <i>M. excavata</i> Schmitz, 1927 | – | – | – | 5 | – | 7 | 5 | 19 | 36 | 0.55 |
| <i>M. flavescens</i> (Wood, 1909) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. flavicans</i> Schmitz, 1935 | 2 | 2 | – | – | – | – | – | – | 4 | 0.06 |
| <i>M. frameata</i> (Schmitz, 1927) | – | 8 | 1 | – | – | 1 | – | 1 | 11 | 0.17 |
| <i>M. fungivora</i> (Wood, 1909) | – | 58 | – | – | – | – | – | 1 | 59 | 0.90 |
| <i>M. furva</i> Schmitz, 1929 | – | 1 | – | 1 | – | 2 | – | 4 | 8 | 0.12 |
| <i>M. fuscipalpis</i> (Lundbeck, 1920) | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. fuscooides</i> Schmitz, 1934 | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. fuscovariana</i> Schmitz, 1933 | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. gartensis</i> Disney, 1985 | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. giraudii</i>-complex | – | 1 | – | – | 24 | 21 | 4 | 5 | 55 | 0.84 |
| <i>M. glabrifrons</i> (Wood, 1909) | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. gregaria</i> (Wood, 1910) | – | 11 | – | – | – | – | – | – | 11 | 0.17 |
| * <i>M. henrydisneyi</i> Durska, 1998 | – | – | 1 | – | – | – | – | 3 | 4 | 0.06 |
| <i>M. hibernans</i> Schmitz, 1934 | 1 | 5 | 2 | 3 | – | – | – | – | 11 | 0.17 |

Table 1, continued

| | | | | | | | | | | |
|---|----------|------------|------------|-----------|------------|------------|-----------|-----------|-------------|--------------|
| <i>M. hilaris</i> Schmitz, 1927 | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. hirticus</i> (Schmitz, 1918) | 3 | 39 | – | – | – | – | – | 1 | 43 | 0.66 |
| <i>M. hyalipennis</i> (Wood, 1912) | – | 1 | – | 11 | 18 | 34 | 3 | – | 67 | 1.02 |
| <i>M. ignobilis</i> (Schmitz, 1919) | – | 12 | 4 | 2 | 1 | – | – | – | 19 | 0.29 |
| <i>M. insons</i> (Lundbeck, 1920) | – | 5 | 11 | 8 | 21 | 72 | – | 2 | 119 | 1.81 |
| <i>M. jameslamonti</i> (Disney, 1995) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. latifrons</i> (Wood, 1910) | – | – | 2 | – | – | 2 | – | – | 4 | 0.06 |
| <i>M. latior</i> Schmitz, 1936 | – | 4 | – | – | – | – | – | – | 4 | 0.06 |
| <i>M. limburgensis</i> (Schmitz, 1918) | – | 46 | – | – | – | – | – | 1 | 47 | 0.72 |
| <i>M. longipalpis</i> (Wood, 1910) | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. lucifrons</i> (Schmitz, 1918) | 4 | 17 | 4 | – | 1 | 1 | – | – | 27 | 0.41 |
| <i>M. lutea</i> (Meigen, 1830) | – | 6 | – | – | 3 | 11 | – | – | 20 | 0.30 |
| <i>M. luteipes</i> (Schmitz, 1918) | 2 | 4 | – | – | – | – | – | – | 6 | 0.09 |
| <i>M. major</i> (Wood, 1912) | – | 1 | – | – | – | – | – | 4 | 5 | 0.08 |
| <i>M. maura</i> (Wood, 1910) | – | 6 | – | – | – | – | – | – | 6 | 0.09 |
| <i>M. meconicera</i> (Speiser, 1925) | – | 3 | 5 | – | – | – | 3 | 8 | 19 | 0.29 |
| <i>M. minor</i> (Zetterstedt, 1848) | – | – | 1 | – | – | – | – | – | 1 | 0.02 |
| <i>M. nasoni</i> (Malloch, 1914) | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. nigra</i> (Meigen, 1830) | – | 1 | 1 | – | – | – | – | – | 2 | 0.03 |
| <i>M. nigriceps</i> (Loew, 1866) | – | 380 | 172 | 50 | 57 | 88 | 49 | 72 | 868 | 13.22 |
| <i>M. obscuripennis</i> (Wood, 1909) | – | – | – | – | – | 3 | – | – | 3 | 0.05 |
| <i>M. oweni</i> Disney, 1988 | – | – | – | – | 6 | 1 | 6 | 2 | 15 | 0.23 |
| <i>M. pectoralis</i> (Wood, 1910) | – | 86 | 2 | 2 | 9 | 1 | 3 | 2 | 105 | 1.60 |
| <i>M. pectorella</i> Schmitz, 1929 | – | 285 | 51 | – | – | – | – | – | 336 | 5.12 |
| <i>M. pleuralis</i> (Wood, 1909) | – | 156 | 941 | 17 | 393 | 301 | 66 | 58 | 1932 | 29.43 |
| <i>M. plurispinulosa</i> (Zetterstedt, 1860) | – | 26 | – | 2 | 3 | 11 | – | 2 | 44 | 0.67 |
| <i>M. propinqua</i> (Wood, 1909) | – | 1 | – | – | – | – | – | – | 1 | 0.02 |
| <i>M. protarsalis</i> Schmitz, 1927 | – | – | – | – | – | 1 | – | – | 1 | 0.02 |
| <i>M. pulicaria</i> -complex | 7 | 40 | 12 | 5 | 141 | 162 | 79 | 62 | 508 | 7.74 |
| <i>M. pumila</i> (Meigen, 1830) | 2 | 116 | 64 | 28 | 1 | 3 | 2 | 1 | 217 | 3.31 |
| <i>M. pusilla</i> (Meigen, 1830) | – | 110 | 50 | 6 | 1 | – | 6 | 4 | 177 | 2.70 |
| <i>M. quadriseta</i> (Schmitz, 1918) | 1 | 16 | 2 | – | – | 2 | 2 | – | 23 | 0.35 |
| <i>M. rubella</i> (Schmitz, 1920) | – | 5 | 1 | – | 1 | 4 | – | – | 11 | 0.17 |
| <i>M. rubescens</i> (Wood, 1912) | – | 2 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. rufipes</i> (Meigen, 1804) | – | – | – | – | 2 | 1 | – | – | 3 | 0.05 |
| <i>M. setulipalpis</i> Schmitz, 1938 | – | 34 | 7 | 5 | 1 | 6 | – | 4 | 57 | 0.87 |
| <i>M. simulans</i> (Wood, 1912) | – | 1 | – | – | – | 1 | – | – | 2 | 0.03 |
| <i>M. sordida</i> (Zetterstedt, 1838) | – | 4 | – | – | – | – | – | – | 4 | 0.06 |
| <i>M. speiseri</i> Schmitz, 1929 | – | – | 1 | – | – | – | – | – | 1 | 0.02 |
| <i>M. spinata</i> (Wood, 1910) | – | 6 | – | – | – | – | – | – | 6 | 0.09 |
| <i>M. spinicincta</i> (Wood, 1910) | 5 | 2 | 1 | 1 | – | – | – | – | 9 | 0.14 |
| <i>M. spinigera</i> (Wood, 1908) | 1 | 1 | – | – | – | – | – | – | 2 | 0.03 |
| <i>M. stichata</i> (Lundbeck, 1920) | – | – | – | – | 1 | – | – | – | 1 | 0.02 |
| <i>M. striolata</i> Schmitz, 1940 | 1 | 8 | 4 | 2 | 2 | 7 | 2 | – | 26 | 0.40 |
| <i>M. subcarpalis</i> (Lundbeck, 1920) | – | – | – | 2 | – | 1 | – | – | 3 | 0.05 |
| <i>M. subpalpalis</i> (Lundbeck, 1920) | – | – | – | 1 | 2 | – | 5 | 2 | 10 | 0.15 |
| <i>M. subpleuralis</i> (Wood, 1909) | – | 14 | – | 3 | 3 | 16 | 7 | 12 | 55 | 0.84 |
| <i>M. sulphuripes</i> (Wood, 1910) | – | 2 | 1 | 3 | – | – | – | – | 6 | 0.09 |
| <i>M. sylvatica</i> (Wood, 1910) | – | 3 | – | 2 | 4 | 2 | 1 | 6 | 18 | 0.27 |
| <i>M. tama</i> (Schmitz, 1926) | – | – | 1 | – | 1 | – | – | – | 2 | 0.03 |
| <i>M. tarsalis</i> (Wood, 1910) | – | 3 | – | – | – | – | – | – | 3 | 0.05 |
| <i>M. unicolor</i> (Schmitz, 1919) | – | – | – | – | 80 | 167 | 17 | 70 | 334 | 5.09 |
| <i>M. variana</i> Schmitz, 1926 | 1 | 10 | 4 | 4 | – | – | – | – | 19 | 0.29 |
| <i>M. woodi</i> (Lundbeck, 1922) | – | – | – | – | – | 2 | 5 | 5 | 12 | 0.18 |
| <i>M. xanthozona</i> (Strobl, 1892) | – | 1 | 6 | 2 | – | – | – | – | 9 | 0.14 |
| <i>Megaselia</i> sp. males | 7 | 76 | 21 | 7 | 30 | 33 | 4 | 7 | 185 | 2.82 |
| *<i>Menoziola obscuripes</i> (Schmitz, 1927) | – | 83 | 70 | – | – | – | – | – | 153 | 2.33 |

Table 1, continued

| | | | | | | | | | | |
|---|----|------|------|-----|-----|------|-----|-----|------|------|
| * <i>Metopina galeata</i> (Haliday, 1833) | – | – | 1 | – | – | – | – | – | 1 | 0.02 |
| <i>Phora atra</i> (Meigen, 1804) | – | – | 5 | – | – | – | – | – | 5 | 0.08 |
| <i>Phora dubia</i> (Zetterstedt, 1848) | 1 | – | – | – | – | – | – | – | 1 | 0.02 |
| * <i>Plectanocnema nudipes</i> (Becker, 1901) | 1 | – | – | – | – | – | – | – | 1 | 0.02 |
| * <i>Pseudacteon formicarum</i> (Verralli, 1877) | – | – | – | 3 | – | – | – | – | 3 | 0.05 |
| <i>Triphleba minuta</i> (Fabricius, 1787) | – | – | – | – | – | 1 | – | – | 1 | 0.02 |
| <i>Triphleba opaca</i> (Meigen, 1830) | 1 | – | – | – | – | – | – | – | 1 | 0.02 |
| Total | 45 | 1991 | 1678 | 207 | 897 | 1081 | 278 | 387 | 6563 | 100 |

selia spp., and were not included in the analysis. All material from this study is deposited at the Swedish Museum of Natural History, Stockholm.

2.2. Study sites

The most common type of habitat before the large wildfire was mature pine forest on bedrock, constituting more than 50% of the total area. Then follows spruce-dominated forest on the slopes and bottoms of the fissure-valleys (30%), wetland (11%), mixed forest (6%), and deciduous forest (2%) (Pettersson 2006).

All traps considered in this study were situated on burnt mature pine forests on bedrock, and the intensity of the fires ranged from weak to medium, with the associated destruction of the humus layer of the soil ranging from moderate to deep. The intensity of fire is strong when the crowns of the trees are dead and weak when these are not affected by the fire and therefore still green (i.e. flame height ranging from weak, moderate, intermediate to strong). Within the major wildfire area, the smoldering fires in the humus destroyed most of the organic soil in some parts (i.e., effect of the soil ranging from weak, moderate, intermediate to strong). Four traps were deployed on three wildfire sites A, B and C (Fig. 1) and were emptied at intervals of 24 to 49 days (for further details see Table 1). In the following, the sites are described in more details.

- Site A: medium-sized (20 ha) and dominated by *Pinus-Betula* tree species. The fire occurred on the 6th of June 1997. One trap (nr 1: samples 27 and 29) was deployed two years after the fire on a spot that had been exposed

to intermediate fire intensity and soil destruction.

- Site B: small-sized (2–3 ha) and dominated by *Pinus-Picea* tree species. The fire occurred on the 1st of June 1999. One trap (nr 2: samples 9 and 10) was deployed 10 days after the fire on a spot that had been exposed to intermediate fire intensity and moderate soil destruction.
- Site C: large-sized (450 ha) and dominated by *Pinus-Betula* tree species. The fire occurred between 1–12 August 1999. Two traps (nr 3 and 4: samples 17–20) were deployed close to each other directly after the fire on spots that had been exposed to intermediate fire intensity and strong soil destruction.

2.3. Data analysis

Analyses were based solely on male individuals, as most females of *Megaselia* spp. and *Phora* spp. are not identifiable to species level, consequently, these have only been identified to genus level. We described the scuttle fly assemblages by dominance structures and the relative abundances of species for each site. We classed species as “dominant” if its relative abundance was $\geq 1\%$.

We compared seasonal changes (phenology) in abundance (n – number of specimens) and species richness (s – number of species). Since the sampling effort varied between periods, we standardized the number of active sampling days to 20 [(actual number of individuals collected/number of days of active sampling) \times 20, calculated in Microsoft Excel 2003] to make the number of individuals comparable. To correct species richness for the influence of different sampling efforts between the samples, we used a non-para-

Table 2. List of species (males) with the number of individuals standardized to 20 trap days (based on Table 1) [(number of individuals collected/number of trap days) × 20 days]. Relative abundance (%) is given for the total sample. Dominant species for the equal sampling effort with dominance ≥ 1% are shown in bold type. *T.S* = total sample. Species with * are new to Sweden.

| Sample | 29 | 27 | 9 | 10 | 17 | 19 | 18 | 20 | <i>T.S</i> | <i>T.S</i> (%) |
|----------------------------------|-------------|--------------|---------------|--------------|--------------|--------------|-------------|--------------|---------------|----------------|
| <i>Anevrina unispinosa</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| * <i>Beckerina umbrimargo</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>Conicera floricola</i> | – | – | 1.08 | – | – | – | – | – | 1.08 | 0.03 |
| <i>Conicera similis</i> | – | – | 1.62 | – | – | – | – | – | 1.62 | 0.05 |
| <i>Conicera tibialis</i> | – | – | 1.08 | – | – | – | – | – | 1.08 | 0.03 |
| <i>Diplonevra nitidula</i> | – | – | 0.54 | – | 12.86 | – | – | 0.95 | 14.35 | 0.40 |
| <i>Hypocerides nearcticus</i> | – | – | – | – | 2.14 | – | – | – | 2.14 | 0.06 |
| <i>Megaselia abdita</i> | – | – | 0.54 | – | – | – | – | – | 0.54 | 0.02 |
| <i>M. aculeata</i> | – | – | – | – | – | – | 0.48 | – | 0.48 | 0.01 |
| <i>M. aequalis</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. affinis</i> | – | 0.41 | 2.70 | 2.27 | – | – | – | – | 5.38 | 0.15 |
| <i>M. albicaudata</i> | – | 2.04 | – | – | – | – | – | – | 2.04 | 0.06 |
| <i>M. albiclava</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. albocingulata</i> | – | 4.08 | 2.70 | 1.36 | 2.86 | 14.29 | 0.48 | 0.48 | 26.24 | 0.73 |
| <i>M. annulipes</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. barbulata</i> | – | – | – | – | – | 0.71 | – | – | 0.71 | 0.02 |
| <i>M. basispinata</i> | – | 5.71 | 10.27 | 0.91 | 2.14 | 6.43 | – | – | 25.47 | 0.71 |
| <i>M. berndseni</i> | – | 0.82 | 2.16 | – | 0.71 | – | – | – | 3.69 | 0.10 |
| <i>M. bifida</i> | – | 0.41 | – | – | 0.71 | 0.71 | – | – | 1.84 | 0.05 |
| <i>M. brevicostalis</i> | – | 51.02 | 100.00 | 7.73 | 34.29 | 46.43 | – | 3.81 | 243.27 | 6.79 |
| <i>M. campestris</i> | – | – | – | 0.45 | – | – | – | – | 0.45 | 0.01 |
| <i>M. ciliata</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. communiformis</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| <i>M. conformis</i> | 1.67 | 0.41 | – | – | – | – | – | – | 2.07 | 0.06 |
| <i>M. consetigera</i> | – | 5.31 | – | – | – | – | – | – | 5.31 | 0.15 |
| <i>M. crassipes</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. curvicapilla</i> | 2.50 | 11.02 | 0.54 | – | – | – | – | – | 14.06 | 0.39 |
| <i>M. differens</i> | – | 2.45 | – | – | – | – | – | – | 2.45 | 0.07 |
| <i>M. discreta</i> | – | 2.04 | – | 0.45 | 2.86 | 0.71 | – | – | 6.07 | 0.17 |
| <i>M. diversa</i> | – | 7.35 | – | – | 6.43 | 13.57 | 3.33 | 8.57 | 39.25 | 1.10 |
| <i>M. drakei</i> | – | 2.45 | – | 0.91 | – | – | – | – | 3.36 | 0.09 |
| <i>M. eccoptomera</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| <i>M. elongata</i> | – | – | 0.54 | – | – | – | – | – | 0.54 | 0.02 |
| <i>M. emarginata</i> | – | 8.16 | – | 0.45 | – | – | – | – | 8.62 | 0.24 |
| <i>M. excavata</i> | – | – | – | 2.27 | – | 5.00 | 2.38 | 9.05 | 18.70 | 0.52 |
| <i>M. flavescens</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. flavicans</i> | 1.67 | 0.82 | – | – | – | – | – | – | 2.48 | 0.07 |
| <i>M. frameata</i> | – | 3.27 | 0.54 | – | – | 0.71 | – | 0.48 | 5.00 | 0.14 |
| <i>M. fungivora</i> | – | 23.67 | – | – | – | – | – | 0.48 | 24.15 | 0.67 |
| <i>M. furva</i> | – | 0.41 | – | 0.45 | – | 1.43 | – | 1.90 | 4.20 | 0.12 |
| <i>M. fuscipalpis</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. fuscoides</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. fuscovariana</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. gartensis</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. giraudii-compl.</i> | 0.41 | – | – | 17.14 | 15.00 | 1.90 | 2.38 | 36.84 | 1.03 | |
| <i>M. glabrifrons</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| <i>M. gregaria</i> | – | 4.49 | – | – | – | – | – | – | 4.49 | 0.13 |
| * <i>M. henrydisneyi</i> | – | – | 0.54 | – | – | – | – | 1.43 | 1.97 | 0.05 |
| <i>M. hibernans</i> | 0.83 | 2.04 | 1.08 | 1.36 | – | – | – | – | 5.32 | 0.15 |
| <i>M. hilaris</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. hirticrus</i> | 2.50 | 15.92 | – | – | – | – | – | 0.48 | 18.89 | 0.53 |
| <i>M. hyalipennis</i> | – | 0.41 | – | 5.00 | 12.86 | 24.29 | 1.43 | – | 43.98 | 1.23 |
| <i>M. ignobilis</i> | – | 4.90 | 2.16 | 0.91 | 0.71 | – | – | – | 8.68 | 0.24 |
| <i>M. insons</i> | – | 2.04 | 5.95 | 3.64 | 15.00 | 51.43 | – | 0.95 | 79.00 | 2.21 |
| <i>M. jameslamonti</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |

Table 2, continued

| | | | | | | | | | | |
|-------------------------------------|-------------|---------------|---------------|--------------|---------------|---------------|--------------|--------------|----------------|--------------|
| <i>M. latifrons</i> | – | – | 1.08 | – | – | 1.43 | – | – | 2.51 | 0.07 |
| <i>M. lator</i> | – | 1.63 | – | – | – | – | – | – | 1.63 | 0.05 |
| <i>M. limburgensis</i> | – | 18.78 | – | – | – | – | – | 0.48 | 19.25 | 0.54 |
| <i>M. longipalpis</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| <i>M. lucifrons</i> | 3.33 | 6.94 | 2.16 | – | 0.71 | 0.71 | – | – | 13.86 | 0.39 |
| <i>M. lutea</i> | – | 2.45 | – | – | 2.14 | 7.86 | – | – | 12.45 | 0.35 |
| <i>M. luteipes</i> | 1.67 | 1.63 | – | – | – | – | – | – | 3.30 | 0.09 |
| <i>M. major</i> | – | 0.41 | – | – | – | – | – | 1.90 | 2.31 | 0.06 |
| <i>M. maura</i> | – | 2.45 | – | – | – | – | – | – | 2.45 | 0.07 |
| <i>M. meconicera</i> | – | 1.22 | 2.70 | – | – | – | 1.43 | 3.81 | 9.17 | 0.26 |
| <i>M. minor</i> | – | – | 0.54 | – | – | – | – | – | 0.54 | 0.02 |
| <i>M. nasoni</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. nigra</i> | – | 0.41 | 0.54 | – | – | – | – | – | 0.95 | 0.03 |
| <i>M. nigriceps</i> | – | 155.10 | 92.97 | 22.73 | 40.71 | 62.86 | 23.33 | 34.29 | 431.99 | 12.07 |
| <i>M. obscuripennis</i> | – | – | – | – | – | 2.14 | – | – | 2.14 | 0.06 |
| <i>M. oweni</i> | – | – | – | – | 4.29 | 0.71 | 2.86 | 0.95 | 8.81 | 0.25 |
| <i>M. pectoralis</i> | – | 35.10 | 1.08 | 0.91 | 6.43 | 0.71 | 1.43 | 0.95 | 46.62 | 1.30 |
| <i>M. pectorella</i> | – | 116.33 | 27.57 | – | – | – | – | – | 143.89 | 4.02 |
| <i>M. pleuralis</i> | – | 63.67 | 508.65 | 7.73 | 280.71 | 215.00 | 31.43 | 27.62 | 1134.81 | 31.70 |
| <i>M. plurispinulosa</i> | – | 10.61 | – | 0.91 | 2.14 | 7.86 | – | 0.95 | 22.47 | 0.63 |
| <i>M. propinqua</i> | – | 0.41 | – | – | – | – | – | – | 0.41 | 0.01 |
| <i>M. protarsalis</i> | – | – | – | – | – | 0.71 | – | – | 0.71 | 0.02 |
| <i>M. pulicaria-comp.</i> | 5.83 | 16.33 | 6.49 | 2.27 | 100.71 | 115.71 | 37.62 | 29.52 | 314.49 | 8.78 |
| <i>M. pumila</i> | 1.67 | 47.35 | 34.59 | 12.73 | 0.71 | 2.14 | 0.95 | 0.48 | 100.62 | 2.81 |
| <i>M. pusilla</i> | – | 44.90 | 27.03 | 2.73 | 0.71 | – | 2.86 | 1.90 | 80.13 | 2.24 |
| <i>M. quadriseta</i> | 0.83 | 6.53 | 1.08 | – | – | 1.43 | 0.95 | – | 10.83 | 0.30 |
| <i>M. rubella</i> | – | 2.04 | 0.54 | – | 0.71 | 2.86 | – | – | 6.15 | 0.17 |
| <i>M. rubescens</i> | – | 0.82 | – | – | – | – | – | – | 0.82 | 0.02 |
| <i>M. rufipes</i> | – | – | – | – | 1.43 | 0.71 | – | – | 2.14 | 0.06 |
| <i>M. setulipalpis</i> | – | 13.88 | 3.78 | 2.27 | 0.71 | 4.29 | – | 1.90 | 26.84 | 0.75 |
| <i>M. simulans</i> | – | 0.41 | – | – | – | 0.71 | – | – | 1.12 | 0.03 |
| <i>M. sordida</i> | – | 1.63 | – | – | – | – | – | – | 1.63 | 0.05 |
| <i>M. speiseri</i> | – | – | 0.54 | – | – | – | – | – | 0.54 | 0.02 |
| <i>M. spinata</i> | – | 2.45 | – | – | – | – | – | – | 2.45 | 0.07 |
| <i>M. spinicincta</i> | 4.17 | 0.82 | 0.54 | 0.45 | – | – | – | – | 5.98 | 0.17 |
| <i>M. spinigera</i> | 0.83 | 0.41 | – | – | – | – | – | – | 1.24 | 0.03 |
| <i>M. stichata</i> | – | – | – | – | 0.71 | – | – | – | 0.71 | 0.02 |
| <i>M. striolata</i> | 0.83 | 3.27 | 2.16 | 0.91 | 1.43 | 5.00 | 0.95 | – | 14.55 | 0.41 |
| <i>M. subcarpalis</i> | – | – | – | 0.91 | – | 0.71 | – | – | 1.62 | 0.05 |
| <i>M. subpalpalis</i> | – | – | – | 0.45 | 1.43 | – | 2.38 | 0.95 | 5.22 | 0.15 |
| <i>M. subpleuralis</i> | – | 5.71 | – | 1.36 | 2.14 | 11.43 | 3.33 | 5.71 | 29.70 | 0.83 |
| <i>M. sulphuripes</i> | – | 0.82 | 0.54 | 1.36 | – | – | – | – | 2.72 | 0.08 |
| <i>M. sylvatica</i> | – | 1.22 | – | 0.91 | 2.86 | 1.43 | 0.48 | 2.86 | 9.75 | 0.27 |
| <i>M. tama</i> | – | – | 0.54 | – | 0.71 | – | – | – | 1.25 | 0.04 |
| <i>M. tarsalis</i> | – | 1.22 | – | – | – | – | – | – | 1.22 | 0.03 |
| <i>M. unicolor</i> | – | – | – | – | 57.14 | 119.29 | 8.10 | 33.33 | 217.86 | 6.08 |
| <i>M. variana</i> | 0.83 | 4.08 | 2.16 | 1.82 | – | – | – | – | 8.90 | 0.25 |
| <i>M. woodi</i> | – | – | – | – | – | 1.43 | 2.38 | 2.38 | 6.19 | 0.17 |
| <i>M. xanthozona</i> | – | 0.41 | 3.24 | 0.91 | – | – | – | – | 4.56 | 0.13 |
| <i>Megaselia</i> sp. males | 5.83 | 31.02 | 11.35 | 3.18 | 21.43 | 23.57 | 1.90 | 3.33 | 101.63 | 2.84 |
| *<i>Menoziola obscuripes</i> | – | 33.88 | 37.84 | – | – | – | – | – | 71.72 | 2.00 |
| * <i>Metopina galeata</i> | – | – | 0.54 | – | – | – | – | – | 0.54 | 0.02 |
| <i>Phora atra</i> | – | – | 2.70 | – | – | – | – | – | 2.70 | 0.08 |
| <i>Phora dubia</i> | 0.83 | – | – | – | – | – | – | – | 0.83 | 0.02 |
| * <i>Plectanocnema nudipes</i> | 0.83 | – | – | – | – | – | – | – | 0.83 | 0.02 |
| * <i>Pseudacteon formicarium</i> | – | – | – | 1.36 | – | – | – | – | 1.36 | 0.04 |
| <i>Triphleba minuta</i> | – | – | – | – | – | 0.71 | – | – | 0.71 | 0.02 |
| <i>Triphleba opaca</i> | 0.83 | – | – | – | – | – | – | – | 0.83 | 0.02 |
| Total | 37.5 | 812.2 | 907.0 | 94.1 | 640.7 | 772.1 | 132.4 | 184.3 | 3580.4 | 100 |

Table 3. Sampling periods and summary values of number of species of the scuttle fly assemblages of the Tyresta Forest after wildfires. *Sobs* = species observed, *Chao1* = Chao1 estimate, *S.D.* = standard deviation. Letters in parenthesis refer to settings in Fig. 1.

| Sample | 29 (A) | 27 (A) | 9 (B) | 10 (B) | 17 (C) | 19 (C) | 18 (C) | 20 (C) | Total |
|--------------|---------------|-----------------|------------------|--------------------|------------------|------------------|---------------|---------------|-------|
| Period | 05.V– 28.V | 28.V– 15.VII | 11.VI– 17.VII | 17.VII– 29.VIII | 13.VIII– 9.IX | 13.VIII– 9.IX | 9.IX– 20.X | 9.IX– 20.X | |
| <i>Sobs</i> | 17 | 80 | 42 | 32 | 33 | 37 | 21 | 29 | 112 |
| Singletons | 8 | 21 | 14 | 6 | 10 | 12 | 3 | 6 | 24 |
| Doubletons | 4 | 10 | 6 | 9 | 3 | 5 | 3 | 6 | 12 |
| <i>Chao1</i> | 25 | 102 | 58 | 34 | 50 | 51 | 23 | 32 | 136 |
| <i>S.D.</i> | 7.5 | 12.8 | 11.7 | 2.3 | 14.8 | 11.2 | 2.3 | 3.2 | 12.9 |

Table 4. Summary values of observed species richness, species richness estimate and trap days from different localities. *Sobs* = species observed, *Chao1* = Chao 1 estimate, *S.D.* = standard deviation.

| | <i>Sobs</i> | <i>Chao1</i> | <i>S.D.</i> | Trap days |
|----------------------------|-------------|--------------|-------------|-------------------------|
| Tyresta Forest | 112 | 136 | 12.96 | 294 |
| Ticino Forest | 63 | 77 | 8.43 | 161 (483 ^a) |
| Białowieża Primeval Forest | 75 | 95 | 11.83 | 358 |
| Tuchola Forest | 77 | 106 | 15.81 | 358 |
| Biała Forest | 52 | 63 | 7.67 | 358 |
| Pisz Forest | 72 | 84 | 8.81 | 118 |

a) 483 trap days including all three traps on each site (table 2 in Prescher *et al.* 2002)

Table 5. Percentage of dominants ($n \geq 1\%$) on each site after wildfires in 1997 and 1999. All sampling was made in 1999. Letters in parenthesis indicate settings in Fig. 1.

| Species | 1997 | | 1999 | | | | | |
|------------------------------------|----------------|----------------|---------------|----------------|----------------|----------------|----------------|----------------|
| | 29 (A) 1999 | 27 (A) 1999 | 9 (B) 1999 | 10 (B) 1999 | 17 (C) 1999 | 19 (C) 1999 | 18 (C) 1999 | 20 (C) 1999 |
| <i>Megaselia pleuralis</i> | – | 7.84 | 56.08 | 8.21 | 43.81 | 27.84 | 23.74 | 14.99 |
| <i>Megaselia nigriceps</i> | – | 19.09 | 10.25 | 24.15 | 6.35 | 8.14 | 17.63 | 18.60 |
| <i>Megaselia pulicaria</i> -compl. | 15.56 | 2.01 | 0.72 | 2.42 | 15.72 | 14.99 | 28.42 | 16.02 |
| <i>Megaselia brevicostalis</i> | – | 6.28 | 11.03 | 8.21 | 5.35 | 6.01 | – | 2.07 |
| <i>Megaselia unicolor</i> | – | – | – | – | 8.92 | 15.45 | 6.12 | 18.09 |
| <i>Megaselia pectorella</i> | – | 14.31 | 3.04 | – | – | – | – | – |
| <i>Megaselia pumila</i> | 4.44 | 5.83 | 3.81 | 13.53 | 0.11 | 0.28 | 0.72 | 0.26 |
| <i>Megaselia pusilla</i> | – | 5.52 | 2.98 | 2.90 | 0.11 | – | 2.16 | 1.03 |
| <i>Megaselia insons</i> | – | 0.25 | 0.66 | 3.86 | 2.34 | 6.66 | – | 0.52 |
| <i>Menozziola obscuripes</i> | – | 4.17 | 4.17 | – | – | – | – | – |
| <i>Megaselia pectoralis</i> | – | 4.32 | 0.12 | 0.97 | 1.00 | 0.09 | 1.08 | 0.52 |
| <i>Megaselia hyalipennis</i> | – | 0.05 | – | 5.31 | 2.01 | 3.15 | 1.08 | – |
| <i>Megaselia diversa</i> | – | 0.90 | – | – | 1.00 | 1.76 | 2.52 | 4.65 |
| <i>Megaselia giraudii</i> -compl. | – | 0.05 | – | – | 2.68 | 1.94 | 1.44 | 1.29 |
| Total | 20.00 | 71.57 | 92.86 | 69.56 | 86.72 | 84.37 | 83.47 | 76.75 |

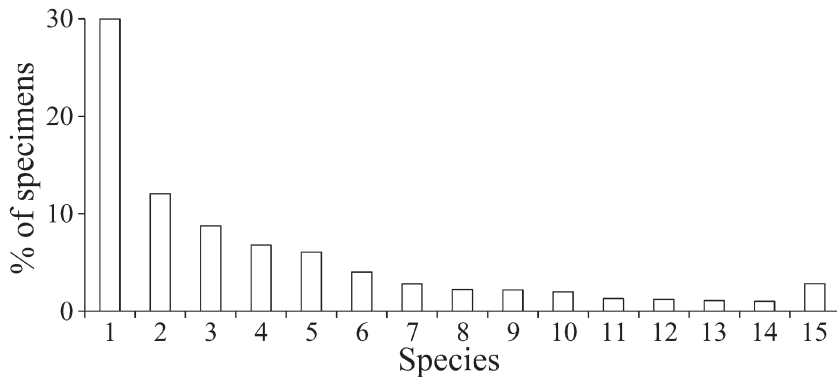


Fig. 2. Percentage of species representing $\geq 1\%$ of scuttle flies collected in 1999, all samples, standardized to a sampling effort of 20 days. Numbers on the x-axis refer to the names of dominant species: 1 – *Megaselia pleuralis*, 2 – *Megaselia nigriceps*, 3 – *Megaselia pulicaria*-complex, 4 – *Megaselia brevicostalis*, 5 – *Megaselia unicolor*, 6 – *Megaselia pectorella*, 7 – *Megaselia pumila*, 8 – *Megaselia pusilla*, 9 – *Megaselia insons*, 10 – *Menoziola obscuripes*, 11 – *Megaselia pectoralis*, 12 – *Megaselia hyalinipennis*, 13 – *Megaselia diversa*, 14 – *M. giraudii* – complex, 15 – *Megaselia* sp. males.

metric abundance-based species richness estimator, *Chao1* (Chao 1984, 1987, Chao & Shen 2003). Species richness estimations and statistics were made in SPADE (Chao & Shen 2003).

3. Results

We identified 12 genera and 112 species from a total of 8,051 individuals (i.e., including both males and females of scuttle flies). Six species were new to Sweden (Tables 1 & 2).

3.1. Assemblage structure

Individuals of species in the genus *Megaselia* (i.e. 7,816 specimens) constituted from 93% to 100% of the scuttle fly assemblage associated with each site. We restricted our analyses to males identified to species level (i.e. 6,563 specimens), which constituted over 80% of all individuals collected (Table 1). The total species richness (identified number of species) of the scuttle fly assemblage of Tyresta in 1999 after the wildfires was rather high ($s = 112$, *Chao 1* = 136, *S.D.* = 12.9, trap days = 294) (Tables 3 & 4).

The numbers of species amongst samples varied from $s = 17$ (sample 29) to $s = 80$ (sample 27),

whereas species richness estimates for the same samples, varied from $s = 25$ to $s = 102$ (Table 3). Fourteen of the 112 species collected at Tyresta were considered to be dominant (sum of abundances of all samples $\geq 1\%$): *Megaselia pleuralis*, *M. nigriceps*, *M. pulicaria*-complex, *M. brevicostalis*, *M. pectorella*, *M. unicolor*, *M. pumila*, *M. pusilla*, *Menoziola obscuripes*, *Megaselia insons*, *M. pectoralis*, *M. hyalinipennis*, *M. diversa* and *M. giraudii*-complex (Table 2; Fig. 2). Dominant species per sample and per site are presented in Table 5.

3.2. Phenology

We observed two peaks of abundance: in the spring/summer (sample 27) (May to June), and in summer/autumn (sample 19) (August to September – the smaller one) (Fig. 3). *Chao 1* estimate slightly altered the number of species compared with observed (lower) number of species (Fig. 3).

The most abundant species of the dominant group (i.e. *Megaselia pleuralis*, *M. nigriceps*, *M. pulicaria*-complex and *M. brevicostalis*) are multivoltine, displaying spring and late summer/autumn activity. The majority of the dominant scuttle fly species in Tyresta with known diet preferences are saprophagous species (Table 6).

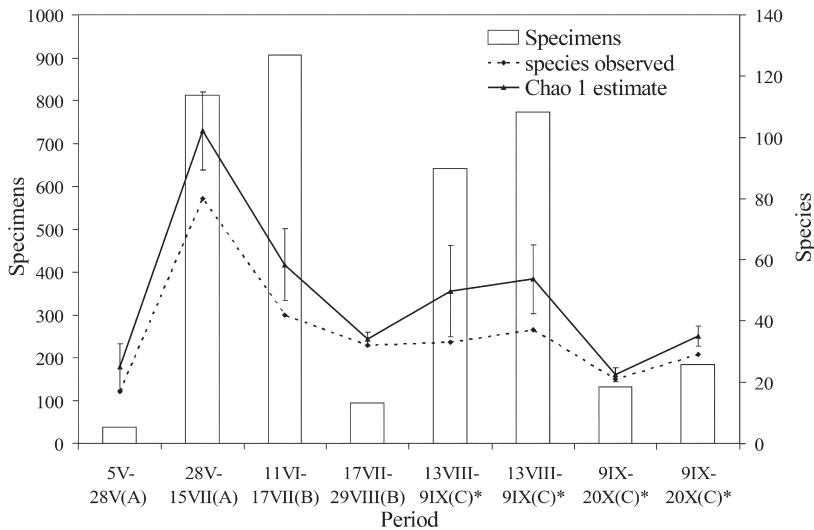


Fig. 3. Seasonal changes in abundance and species richness (number of species based on observed and estimated numbers – *Chao1*) of adult scuttle flies caught in Malaise traps in 1999. Error bars signify standard deviation. Letters in parenthesis refer to sites in Figure 1. *: same time periods due to use of two traps on the same location (double traps).

Of the species listed in Tables 1 & 2 14.0% are known to be saprophages, 16.7% fungivores, 7% parasitoids or specialized predators and 62.3% of unknown larval habits.

4. Discussion

Our study reconfirms the results of previous work showing that *Megaselia* species make a major contribution to the assemblages of Phoridae, especially after disturbance or stress (Disney 2004,

Durska 1996, 2001, 2006, 2009). The same result was obtained by Prescher *et al.* (2002) using emergence traps located in the chestnut belt following a wildfire in the Swiss Alps where *Megaselia* accounted for over 90% of all Phoridae sampled.

One major, surprising difference was the absence of *Megaselia verralli* in our Tyresta samples. This species of unknown biology was a eudominant in the Ticino Forest and also a eudominant and a characteristic species of the pine plantations in the Polish Lowland: Biało-

Table 6. Trophic-group and male activity of the dominant species ($n \geq 1$, equal sampling effort) of scuttle flies in the Tyresta Forest after wildfires with percent number of collected specimens. ? = unknown (or probable) diet of larvae.

| Species activity | % | Trophic groups | Time peaks of male |
|-------------------------------------|-------|---------------------|--------------------|
| <i>Megaselia nigriceps</i> | 12.07 | Saprophagous | V–VII, VIII–IX |
| <i>Megaselia pulicaria</i> -complex | 8.78 | Polysaprophagous | V–VII, VIII–IX |
| <i>Megaselia brevicostalis</i> | 6.79 | Polysaprophagous | V–VII, VIII–IX |
| <i>Megaselia unicolor</i> | 6.08 | Saprophagous | VIII–IX |
| <i>Megaselia pectorella</i> | 4.02 | Saprophagous | V–VII |
| <i>Megaselia pumila</i> | 3.31 | ? | V–VII |
| <i>Megaselia pusilla</i> | 2.81 | ? | V–VII |
| <i>Megaselia insons</i> | 2.21 | ? | VIII–IX |
| <i>Menoziola obscuripes</i> | 2.00 | Parasitoids on ants | V–VII |
| <i>Megaselia pectoralis</i> | 1.30 | Saprophagous | V–VII |
| <i>Megaselia hyalipennis</i> | 1.23 | Fungiphagous | VIII–IX |
| <i>Megaselia diversa</i> | 1.10 | Saprophagous (?) | V–VII, III–IX |
| <i>Megaselia giraudii</i> -complex | 1.03 | Polysaprophagous | V–VII, VIII–IX |

wieża Forest, Tuchola Forest and Biała Forest. In the Ticino Forest, Phoridae were collected using emergence traps, but by yellow pan traps in pine plantations of the Polish Lowland (Durska 2001, 2006, 2009, Prescher *et al.* 2002). *M. albo-cingulata*, which belongs to the *M. verralli* group, was caught at all three wildfire localities in Tyresta Forest and is a dominant species in samples 10 and 19 (Table 5). This species (*M. albo-cingulata*) has not been found in the Ticino Forest but is known from wetland and grassland habitats in Switzerland (only 2 specimens, collected using window traps) (Prescher *et al.* 2000). One possibility is that *M. verralli* has soil-dwelling larvae and the emergence trap method could be more efficient and/or that this species is rare this far north in the Palaearctic Region.

The reason for differences in dominant species and species numbers (at least some of them) caught on different localities (i.e., Ticino Forest, pine plantations of the Polish Lowland) may be that different collecting methods are prone to catch different species, which in turn could affect the assemblage structure. For example, the emergence traps collect a group of species that hatch from a small spot on the ground, whereas a Malaise trap is designed to catch species flying around exhibiting positive phototaxis. However, comparisons of different types of trap techniques applied in collecting phorids have rarely been attempted, and those that have been made showed different results and therefore are difficult to assess, see for example Disney *et al.* (1982), Frouz (1999), Brown & Feener (1995), Brown (1996), Hancock & Ward 1996, Weber & Schiegg (2001). Other aspects affecting species and number of species caught are the number of trap days and collecting periods. i.e., more trap days will catch more species and vice versa, and different collecting periods may cover different weather conditions, i.e., days with good or bad weather.

We found that six of the fourteen dominant species of the megadiverse genus *Megaselia* (*M. pleuralis*, *M. nigriceps*, *M. pulicaria*-complex, *M. brevicostalis*, *M. unicolor* and *M. pumila*) were also the dominants in the Ticino Forest and in the pine plantations of the Polish Lowland (Durska 2001, 2006, 2009, Prescher *et al.* 2002). Adding the hurricane of the Pisz Forest in 2002 (Durska 'in prep.'), the dominants of the dis-

turbed habitats mentioned above were *M. pleuralis*, *M. pulicaria*-complex and *M. brevicostalis*. Thus, we suggest that these three species may be considered to be "phorid disturbance indicators". These species are probably dominants (species of relative abundance $\geq 1\%$) in disturbed areas in general, as they are good colonizers, tolerant to abiotic stress, and generalists with a multivoltine life cycle, which has been discussed before by Durska (2001, 2006, 2009) and Prescher *et al.* (2002). Thus, it is possible that these species benefit from habitat heterogeneity followed by various disturbance sources like cutting, wildfires or wind throw, which may create very similar conditions for habitat-insect interactions (Simberloff 1988, Saunders *et al.* 1991, Tschamtkke & Brandl 2004).

Tyresta Forest was the locality with the highest observed species richness (Table 4; these are only preliminary results due to problems mentioned before) and estimated species richness compared with the Ticino Forest, the pine plantations (after clear-cutting) of the Białowieża Forest, the Tuchola Forest, the Biała Forest or the Pisz Forest after windthrow in 2002 (Durska 2006, 2009 and 'in prep.', Prescher *et al.* 2002).

High species richness in wildfire localities during early succession has several explanations. Catastrophic events such as fires and storms create sun-exposed coarse woody debris (CWD). CWD serves as microhabitats for many plants and animals species. In turn, the clearcut harvesting removes most of the stand's biomass (Bouget & Duelli 2004, Price *et al.* 1998).

Species associated with disturbed areas have good dispersal power and may colonize a burnt area from the surrounding patches (Southwood 1962). Some flies are early colonizers attracted to the burnt habitat by smoke and/or heat. However, only a few studies have dealt with assemblages of Diptera in wildfire sites, except for some studies of the phenomenon of some species building up swarms in smoke. This is known for species of *Microsania* (Platypezidae) from nemoral forests in the Palaearctic Region (Belgium, England) (Chandler 1978). *Hormopeza* (Empididae) has been reported to hunt for *Microsania* in burnt areas. On the major fire-site from 1999 in the Tyresta Forest, two smoke-attraction experiments were conducted during the middle of the sum-

mers of 2000 and 2002. These experiments showed that swarms of *Microsania* also occurred in hemiboreal forests in the Palearctic Region and produced catches of *Microsania* and *Hormopeza* (Ahnlund et al. 2006). No scuttle flies were caught during the experiments. However, such experiments should in the future be conducted also in the spring and autumn, during the peak activity periods of scuttle flies.

The effect of fire on the humus layer within the large wildfire site was strong in many places, leaving only the mineral layer of the soil, making evaporation strong. Shelter created by the root systems of the many toppled-over trees, especially old *Pinus*, might have created a microclimate more suitable for scuttle flies than expected, especially compared with the much more common scenario of moderately burned sites where most trees are standing for a long time. Even though phorids were numerous, Sciaridae were more than twice as abundant (Menzel & Viklund 'in prep.') and these lesser fungus gnats might have developed in the mycelia in the soil and in the fruiting bodies of the pioneering fungi (Ascomycetes). Consequently, it is possible that many species of scuttle flies are predators/parasitoids on the very abundant Sciaridae. For example, *M. obscuripennis* caught in the survey is a parasitoid on Sciaridae (Disney 1994).

The autumn rise and fall in abundances and richness of *Megaselia* species is probably connected with the abundance of fungi, directly or via fungi dependent prey species/hosts (Disney 1994, Durska 1996, 2001, 2006, 2009).

Fire and other disturbances, such as clear-cutting of mature stands or wind throw initiate secondary succession of the ecosystem and are "the main source" of habitat heterogeneity. It probably affects diversity and abundance of Phoridae. Many scuttle fly species are known to be very sensitive to changes in their habitat, especially to climatic conditions (Folgarait et al. 2007). During dry years, *M. brevicostalis* reaches the abundance peak much earlier than in years when humidity is typical (Disney et al. 1981). The open-area species of scuttle fly assemblages respond in a parallel way to different disturbances. The same response to disturbances (clear-cutting, grazing and burning) has also been found for spider and carabid fauna (Coddington et al. 1996, Gibson et

al. 1992, Zulka et al. 1997, Moretti et al. 2002, Fernández & Salgado Costas 2004, Huber & Baumgarten 2005).

5. Conclusions

Our investigation showed that there was high species diversity and abundance of scuttle flies after the wildfires. Our results suggest that the scuttle fly assemblages react in a resilient way to environmental disturbance/stress and that Phoridae could be a prime candidate for use in conservation evaluation exercises, in spite of the imperfect knowledge of the alpha taxonomy of this family (Manson et al. 2007, Disney & Durska 2008). It is in turn possible for Phoridae – as a resilient group of very high ecological diversity – to be used in the monitoring of changes in terrestrial habitats following disturbances, or else as indicators within different ecological levels of organization (Durska 2009). Further studies of this are needed.

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