Influence of mowing measures on carabid beetle fauna (Coleoptera: Carabidae) in a post-agricultural area

Abstract

Background and purpose: Some agricultural practices are considered to be useful tools in biodiversity conservation. Therefore, carabid beetles were collected on post-agricultural fallow land in Western Poland in order to study the impact of mowing treatment.

Materials and Methods: Following a “before-after-control-impact” (BACI) study design over the period of two years (2013 – 2014), standard arrays of pitfall traps were installed on six study sites, of which three were treated by mowing at the beginning of July in the second year of study. The influence of this treatment was analysed statistically with respect to the most frequently collected species, selected ecological traits, as well as the mean individual biomass of the carabid assemblages (MIB). Additionally, Non-metric Multidimensional Scaling (NMDS) was carried out.

Results: A total of 1995 individuals belonging to 40 species were collected, with species numbers ranging from 11 to 21 and numbers of individuals ranging from 76 to 278 in the samples. Although some species reacted significantly to the mowing treatment and numbers of individuals of forest species significantly decreased on the treatment sites, in general rather weak effects were observed as a result of the mowing measures. The weak effects of the mowing measures may be explained by the low cutting intensity (only once a year) and differences in environmental conditions between the years of study.

Conclusions: The results of the study are assumed to be useful in the context of planning mowing measures in order to conserve biological diversity. Yet, the results also underline the importance of long-term studies.

INTRODUCTION

Intensification of agriculture is assessed to be a main driver of loss in biological diversity (1, 2, 3, 4). To counteract this tendency, different strategies have been proposed. Positive effects have been proven for the establishment of botanically diverse field margin strips (5, 6). Setting aside agricultural fields has been considered as a measure to enhance biological diversity, too. However, while several studies provide evidence for positive influence of fallow ground (e.g. 7, 8, 9, 10), some studies also report on its negative impact on biodiversity (e.g. 11). According to Burel and Baudry (12), predictions of the effects of land abandonment are difficult and opportunities offered by land abandonment differ with regard to the species. In contrast, various traditional agricultural practices are considered to be useful tools in biodiversity conservation (e.g. 13) and have been integrated into agri-environmental schemes and nature-reserve management (e.g. 14). Amongst such practices are grazing and mowing, the latter as a surrogate for grazing (13, 14). These mea-
sures influence the structure of grassland vegetation, which is a product of the interplay between successional processes and management. The structure of grassland vegetation, however, is crucial for maintaining arthropod diversity (13).

Among arthropods, carabid beetles have an important position in agricultural systems, such as biological control agents on agricultural pests (15) or seed predators of weeds (16, 17). Carabids react to management practices in grassland habitats (18) and have a potential for indicating environmental variation (19). An approach often used is to study changes in ecological traits of the carabid assemblages, e.g. the reproduction period or habitat preferences (19). The mean individual biomass of Carabidae (MIB) has been proposed as an indicator of the stage of succession of a habitat (20, 21). This method assumes an ongoing process of succession with which the MIB of carabids increases. MIB was established in order to assess forest ecosystems in Poland (20). However, this method has also been used in the context of assessing recovery processes in Mediterranean ecosystems or studying forest patch isolation (22, 23). In addition, MIB has been shown to react sensitively to management strategies on post-agricultural areas (24).

In order to improve our knowledge about the impact of mowing measures on carabid beetles we initiated the presented study in a research area of post-agricultural land in Western Poland. The impact of mowing measures on carabid beetles in this area has been studied before on different study plots (e.g. 24, 25). However, the presented paper is the first study using a “before-after-control-impact” (BACI) design (26), applying mowing to the treatment sites in the second year of the study. By comparing carabid data on treatment sites and control sites we wanted to test the following hypotheses:

1) Species react to the mowing measures, expressed as significant differences in catch numbers on the treatment sites, but not on the control sites, between the first and the second year of study.

2) Numbers of individuals of species characterised by specific ecological traits will change significantly on the treatment sites, but not on the control sites.

3) MIB as an indicator of the stage of succession decreases significantly on the treatment sites, but not on the control sites, from the first to the second year of study.

4) Carabid assemblages of mown sites differ from assemblages of unmown sites.

MATERIAL AND METHODS
Study sites and field methods

For our study, we selected the research area „Krzywda,” an area of more than 170 ha including agricultural fields, abandoned farmland, wet habitats and forest sites (27). The study was carried out in 2013 – 2014 on post-agricultural fallow land abandoned from crop production for 21 years in 2013 (27). It followed a „before-after-control-impact” (BACI) design with three treatment sites and
three control sites. Thus, six study sites (B1 – B6) of 50 m x 50 m were established, three of which (B1, B4, B5) were mown with biomass removal on July 5th, 2014, while the remaining three (B2, B3, B6) were left untreated (Fig. 1).

Carabids were collected using pitfall traps from mid-May to mid-September. Traps were glass jars topped with a funnel (upper diameter of about 10 cm) set flush with the soil surface. A roof was suspended a few cm above the funnel and 50 ml ethylene glycol was used as a killing agent and preservative. The traps were checked every 2-3 weeks for proper functioning. If necessary, ethylene glycol was refilled. Three traps (distance 3 m) were installed in the center of each study site. The results of the three traps were pooled to one sample for each study site.

All collected specimens were determined to the species level. Nomenclature follows Freude et al. (28).

Statistical methods

For each species, the total number of individuals per study site was calculated.

The recorded species were classified with respect to their habitat preferences (species characteristic for open habitats, eurytopic species, species characteristic for forests) and breeding type (spring breeders, autumn breeders) based on the literature (28, 29, 30, 31, 32, 33, 34). For each of these ecological traits, the total number of collected individuals per study site was calculated.

For each study site, we also computed MIB values as an indicator of succession. MIB is calculated by summing up the biomass of all carabids in a sample and subsequently dividing it by the number of specimens caught. Biomass values for the species recorded are those cited by Szyszko (20) or obtained using the term by Szyszko (35) which describes the relationship between the body length of a single carabid individual (a) and its biomass (b):

\[ \ln y = -8.92804283 + 2.55549621 \times \ln x \]

For further statistical analysis, we selected species, which were collected with at least 150 individuals, and looked at the ecological traits (species characteristic for open habitats, eurytopic species, species characteristic for forests, spring breeders, autumn breeders) and MIB values calculated for the study sites. Differences in the numbers of individuals of the selected species, numbers of individuals per ecological trait, and MIB values between 2013 and 2014 on the treatment sites and the control sites, this decrease was more pronounced with a decrease in MIB observed on both the treatment sites and the control plots between 2013 and 2014 (Tab. 2). Note-worthy, with the exception of Amara aenea for all species the same tendency was observed on treatment and control sites; i.e. if the mean number increased on treatment sites from 2013 to 2014 then it also increased on the control sites and vice versa. None of the studied ecological traits (species characteristic for open habitats, eurytopic species, species characteristic for forests, spring breeders, autumn breeders) displayed a significant difference between the years of study on the control sites, but on the treatment sites the number of individuals of forest species was significantly lower in 2014 (Tab. 2). MIB values did not decrease significantly as a result of mowing. However, with a decrease in MIB observed on both the treatment and the control sites, this decrease was more pronounced on the treatment sites (Tab. 2).

NMDS (Fig. 2) did not reveal a clear separation of the samples collected on mown study sites (B1, B4, B5 in 2014) from the samples collected on unmown study sites. The stress for the first two ordination axes was low (0.1548).

### Tab. 1: Type of treatment, number of species, and number of individuals per study site in 2013 and 2014.

<table>
<thead>
<tr>
<th>Study site</th>
<th>Site type/treatment</th>
<th>Species</th>
<th>Individuals</th>
</tr>
</thead>
<tbody>
<tr>
<td>2013</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B1</td>
<td>Treatment site/unmown</td>
<td>20</td>
<td>182</td>
</tr>
<tr>
<td>B2</td>
<td>Control site/unmown</td>
<td>17</td>
<td>232</td>
</tr>
<tr>
<td>B3</td>
<td>Control site/unmown</td>
<td>17</td>
<td>151</td>
</tr>
<tr>
<td>B4</td>
<td>Treatment site/unmown</td>
<td>17</td>
<td>235</td>
</tr>
<tr>
<td>B5</td>
<td>Treatment site/unmown</td>
<td>21</td>
<td>160</td>
</tr>
<tr>
<td>B6</td>
<td>Control site/unmown</td>
<td>16</td>
<td>82</td>
</tr>
<tr>
<td>2014</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>B1</td>
<td>Treatment site/mown</td>
<td>17</td>
<td>141</td>
</tr>
<tr>
<td>B2</td>
<td>Control site/unmown</td>
<td>18</td>
<td>158</td>
</tr>
<tr>
<td>B3</td>
<td>Control site/unmown</td>
<td>13</td>
<td>76</td>
</tr>
<tr>
<td>B4</td>
<td>Treatment site/mown</td>
<td>11</td>
<td>147</td>
</tr>
<tr>
<td>B5</td>
<td>Treatment site/mown</td>
<td>21</td>
<td>153</td>
</tr>
<tr>
<td>B6</td>
<td>Control site/unmown</td>
<td>18</td>
<td>278</td>
</tr>
</tbody>
</table>

RESULTS

A total of 1995 individuals belonging to 40 species (Appendix 1) were collected (Tab. 1). Species collected with at least 150 individuals were Harpalus rubripes (309), Harpalus tardus (238), Harpalus rubripes (220), Calathus erratus (163), Amara aenea (162), and Pterostichus niger (153).

Among the six most dominant species only Pterostichus niger and Harpalus rubripes reacted to mowing the treatment sites with significant changes in catch numbers. Both species showed significant lower numbers in 2014 as compared to 2013. For none of the studied species a significant difference in catch numbers were observed on the control plots between 2013 and 2014 (Tab. 2). Noteworthy, with the exception of Amara aenea for all species the same tendency was observed on treatment and control sites; i.e. if the mean number increased on treatment sites from 2013 to 2014 then it also increased on the control sites and vice versa. None of the studied ecological traits (species characteristic for open habitats, eurytopic species, species characteristic for forests, spring breeders, autumn breeders) displayed a significant difference between the years of study on the control sites, but on the treatment sites the number of individuals of forest species was significantly lower in 2014 (Tab. 2). MIB values did not decrease significantly as a result of mowing. However, with a decrease in MIB observed on both the treatment and the control sites, this decrease was more pronounced on the treatment sites (Tab. 2).
**Tab. 2:** Mean values and standard error (SE) of catch sizes of the most frequently collected species, of numbers of individuals belonging to certain ecological groups, and of MIB values on the study sites in 2013 and 2014. Significance refers to comparisons between the respective samples using the Mann-Whitney U test.

<table>
<thead>
<tr>
<th>Species – year</th>
<th>n</th>
<th>Mean ± SE</th>
<th>Significance</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Treatment sites</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>H. rufipes</em> – 2013</td>
<td>3</td>
<td>40.00 ± 5.51</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>H. rufipes</em> – 2014</td>
<td>3</td>
<td>23.33 ± 4.41</td>
<td></td>
</tr>
<tr>
<td><em>H. tardus</em> – 2014</td>
<td>3</td>
<td>11.67 ± 4.70</td>
<td></td>
</tr>
<tr>
<td><em>H. rubripes</em> – 2013</td>
<td>3</td>
<td>28.0 ± 1.15</td>
<td>p &lt; 0.05</td>
</tr>
<tr>
<td><em>H. rubripes</em> – 2014</td>
<td>3</td>
<td>11.67 ± 4.98</td>
<td></td>
</tr>
<tr>
<td><em>C. erratus</em> – 2013</td>
<td>3</td>
<td>15.33 ± 2.19</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>C. erratus</em> – 2014</td>
<td>3</td>
<td>18.00 ± 8.62</td>
<td></td>
</tr>
<tr>
<td><em>A. aenea</em> – 2013</td>
<td>3</td>
<td>2.33 ± 1.33</td>
<td></td>
</tr>
<tr>
<td><em>A. aenea</em> – 2014</td>
<td>3</td>
<td>1.33 ± 1.33</td>
<td></td>
</tr>
<tr>
<td><em>Pt. niger</em> – 2013</td>
<td>3</td>
<td>15.00 ± 1.15</td>
<td>p &lt; 0.05</td>
</tr>
<tr>
<td><em>Pt. niger</em> – 2014</td>
<td>3</td>
<td>6.67 ± 2.19</td>
<td></td>
</tr>
<tr>
<td><strong>Open habitats</strong> – 2013</td>
<td>3</td>
<td>112.33 ± 5.17</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Open habitats</strong> – 2014</td>
<td>3</td>
<td>89.33 ± 11.62</td>
<td></td>
</tr>
<tr>
<td><strong>Eurytopic</strong> – 2013</td>
<td>3</td>
<td>64.33 ± 20.37</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Eurytopic</strong> – 2014</td>
<td>3</td>
<td>51.00 ± 12.34</td>
<td></td>
</tr>
<tr>
<td><strong>Forests</strong> – 2013</td>
<td>3</td>
<td>15.67 ± 1.45</td>
<td>p &lt; 0.05</td>
</tr>
<tr>
<td><strong>Forests</strong> – 2014</td>
<td>3</td>
<td>6.67 ± 2.19</td>
<td></td>
</tr>
<tr>
<td><strong>Spring</strong> – 2013</td>
<td>3</td>
<td>94.00 ± 21.70</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Spring</strong> – 2014</td>
<td>3</td>
<td>70.67 ± 17.82</td>
<td></td>
</tr>
<tr>
<td><strong>Autumn</strong> – 2013</td>
<td>3</td>
<td>98.33 ± 3.67</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Autumn</strong> – 2014</td>
<td>3</td>
<td>76.33 ± 20.67</td>
<td></td>
</tr>
<tr>
<td><strong>MIB</strong> – 2013</td>
<td>3</td>
<td>78.53 ± 5.66</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>MIB</strong> – 2014</td>
<td>3</td>
<td>63.90 ± 8.36</td>
<td></td>
</tr>
<tr>
<td><strong>Control sites</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>H. rufipes</em> – 2013</td>
<td>3</td>
<td>26.33 ± 8.69</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>H. rufipes</em> – 2014</td>
<td>3</td>
<td>13.33 ± 5.36</td>
<td></td>
</tr>
<tr>
<td><em>H. tardus</em> – 2013</td>
<td>3</td>
<td>27.67 ± 17.29</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>H. tardus</em> – 2014</td>
<td>3</td>
<td>18.67 ± 6.94</td>
<td></td>
</tr>
<tr>
<td><em>H. rubripes</em> – 2013</td>
<td>3</td>
<td>22.00 ± 6.11</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>H. rubripes</em> – 2014</td>
<td>3</td>
<td>11.67 ± 3.53</td>
<td></td>
</tr>
<tr>
<td><em>C. erratus</em> – 2013</td>
<td>3</td>
<td>8.00 ± 3.79</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>C. erratus</em> – 2014</td>
<td>3</td>
<td>13.00 ± 12.00</td>
<td></td>
</tr>
<tr>
<td><em>A. aenea</em> – 2013</td>
<td>3</td>
<td>10.00 ± 5.86</td>
<td></td>
</tr>
<tr>
<td><em>A. aenea</em> – 2014</td>
<td>3</td>
<td>40.33 ± 34.89</td>
<td></td>
</tr>
<tr>
<td><em>Pt. niger</em> – 2013</td>
<td>3</td>
<td>17.00 ± 14.57</td>
<td>n.s.</td>
</tr>
<tr>
<td><em>Pt. niger</em> – 2014</td>
<td>3</td>
<td>12.33 ± 7.22</td>
<td></td>
</tr>
<tr>
<td><strong>Open habitats</strong> – 2013</td>
<td>3</td>
<td>88.33 ± 15.21</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Open habitats</strong> – 2014</td>
<td>3</td>
<td>108.33 ± 54.36</td>
<td></td>
</tr>
<tr>
<td><strong>Eurytopic</strong> – 2013</td>
<td>3</td>
<td>49.33 ± 20.63</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Eurytopic</strong> – 2014</td>
<td>3</td>
<td>49.67 ± 13.35</td>
<td></td>
</tr>
<tr>
<td><strong>Forests</strong> – 2013</td>
<td>3</td>
<td>17.33 ± 14.44</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Forests</strong> – 2014</td>
<td>3</td>
<td>12.67 ± 7.51</td>
<td></td>
</tr>
<tr>
<td><strong>Spring</strong> – 2013</td>
<td>3</td>
<td>93.00 ± 25.36</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Spring</strong> – 2014</td>
<td>3</td>
<td>117.67 ± 39.91</td>
<td></td>
</tr>
<tr>
<td><strong>Autumn</strong> – 2013</td>
<td>3</td>
<td>62.00 ± 19.00</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>Autumn</strong> – 2014</td>
<td>3</td>
<td>53.00 ± 18.90</td>
<td></td>
</tr>
<tr>
<td><strong>MIB</strong> – 2013</td>
<td>3</td>
<td>67.53 ± 11.81</td>
<td>n.s.</td>
</tr>
<tr>
<td><strong>MIB</strong> – 2014</td>
<td>3</td>
<td>59.93 ± 9.93</td>
<td></td>
</tr>
</tbody>
</table>
Boer disguise the effects caused by mowing. According to den
tation of carabid assemblages, too. This may to some degree
the study years might have had an influence on the forma-
trol sites, differences in environmental conditions between
for almost all dominant species on the treatment and con-
sect to forest species on the treatment sites, Tab. 2). Reacted most sensitively (significant difference with re-
analysing the ecological traits indicate that forest species
fected by the mowing measures. However, the results of
species breeding later in the year were particularly af-
tumn species (possibly unstable)." This may indicate that
spring and autumn
33).

The weak effects of the mowing measures may be ex-
explained by the assumption that mowing only once at the
beginning of July was not sufficient to significantly over-
come environmental characteristics of the study sites. Pterostichus niger is known to be active in autumn (e.g. 30,
and Harpalus rubripes is mentioned to breed both in
spring and autumn (33). Larsson (29) describes it as „au-
tumn species (possibly unstable).” This may indicate that
species breeding later in the year were particularly af-
fected by the mowing measures. However, the results of
analysing the ecological traits indicate that forest species
reacted most sensitively (significant difference with re-
pect to forest species on the treatment sites, Tab. 2).

Since the basic trends between the years were the same
for almost all dominant species on the treatment and con-
trol sites, differences in environmental conditions between
the study years might have had an influence on the forma-
tion of carabid assemblages, too. This may to some degree
disguise the effects caused by mowing. According to den
Boer (e.g. 38, 39), such stochastic processes will necessar-
ly lead to density fluctuations of individual species on
fixed sampling plots. As a conclusion, long-term sampling
under varying conditions is recommended to precisely
identify the indicator potential of individual species (19).

Intensification of mowing measures (e.g. mowing twice
a year) may lead to more pronounced differences. Kitaha-
ra et al. (40), studying grasslands being mown once up to
3 – 4 times, observed a highly significant decrease in but-
terfly species numbers with increasing disturbance inten-
sity. However, the highest carabid diversity was observed
for medium management intensity in grasslands managed
with different intensities (41). Accordingly, Mazalová et
al. (42) reported the highest species richness of butterflies
and beetles, at a larger time scale, in grasslands mown
once a year, especially when applying a combined regime
of mowing and grazing. According to Morris (13), Cole-
optera in general are more robust to cutting measures than
other arthropods (i.e. Auchenorrhyncha).

The results of this study are assumed to be useful when
planning mowing measures in order to conserve biological
diversity. Please note that the effect of cutting treat-
ment may vary and therefore the timing and frequency of
this management method should be varied (13). Mowing
should also be put into context with other agricultural
practices which might influence species assemblages, e.g.
soil tillage (9, 43). Our results underline the importance
of long-term studies in order to fully understand the reac-
tion of individual species to management measures and
to define their potential as ecological indicators.

ACKNOWLEDGEMENTS

The authors thank two anonymous reviewers and Lu-
cija Šerić Jelaska for valuable comments on the manu-
script and K. Hannig for confirming the identification of
beetles. This paper is communication No. 477 of the
Laboratory of Evaluation and Assessment of Natural Re-
sources, Warsaw University of Life Sciences – SGGW.

REFERENCES

Agricultural intensification and ecosystem properties. Science 277:
504-509 http://dx.doi.org/10.1126/science.277.5325.504
2. KREBS J R, WILSONJ D, BRADBURY R B, SIRWARDENA
G M 1999 The second silent spring? Nature 400: 611-612
3. DONALD P F, SANDERSON F J, BURFIELD I J, VAN BOM-
MEL T, F P J 2006 Further evidence of continent-wide impacts of
agricultural intensification on European farmland birds, 1990–
http://dx.doi.org/10.1016/j.agee.2006.02.007
4. WATT A D, BRADSHAW R H W, YOUNG J, ALARD D, BOL-
GER T, CHAMBERLAIN D, FERNÁNDEZ-GONZÁLEZ F,
FULLER P, GURREAA P, HENLE K, JOHNSON R, KORSÓ Z,
LAVELLE P, NIESEM, NOWICKI P, REBANEM, SCHEI-
DEGGER C, SOUSA J P, VAN SWAAYC, VANBERGEN A
2007 Trends in biodiversity in Europe and the Impact of land-use
5. THOMAS CF G, MARSHALL E J P 1999 Arthropod abundance
and diversity in differently vegetated margins of arable Fields. Ag-
ric Ecosyst Environ 72: 131-144
http://dx.doi.org/10.1016/S0167-8809(98)00169-8
Influence of mowing measures on carabid beetle fauna

A. Schwerk and M. A. Kitka


8. DEENDEREN K, BOSMANS R 1998 Ground beetles (Coleoptera, Carabidae) on set-aside fields in the Campine region and their importance for nature conservation in Flanders (Belgium). Biodivers Conserv 7: 1485-1493 http://dx.doi.org/10.1023/A:100881313102410


10. HOLLAND J, BIRKETT T, BEGBIE M, SOUTHWAY S, THOMAS C F G 2003 The spatial dynamics of predatory arthropods and the importance of crop and adjacent margin habitats. IOBC wprs Bull 26: 65-70


15. HUMPHREYS I C, MOWAT, D J 1994 Effects of some organic treatments on predators (Coleoptera: Carabidae) of Cabbage Root Fly, Delia radicum (L.) (Diptera, Anthomyiidae), and on an alternative prey species. Pedobiologia 38: 513-518


19. KOIVULA M 2001 Useful model organisms, indicators, or both? Ground beetles (Coleoptera, Carabidae) reflecting environmental conditions. ZooKeys 100: 287-317 http://dx.doi.org/10.3897/zookeys.100.1533


23. ŠЕРIČ JELASKA L, DURБEŠIĆ P 2009 Comparison of the body size and wing form of carabid species (Coleoptera: Carabidae) between isolated and continuous forest habitats. Ann Soc Entomol Fr (n.s.) 45: 327-338

24. SCHWERKA, SZYSZKO J 2009 Distribution and spatial preferences of carabid species (Coleoptera: Carabidae) in a forest-field landscape in Poland. Baltic J Coleopterol 9: 5-15

25. BLASZKIEWICZ M, SCHWERK A 2013 Carabid beetle (Coleoptera: Carabidae) diversity in agricultural and post-agricultural areas in relation to the surrounding habitats. Baltic J Coleopterol 13: 15-26


34. HURKA K 1996 Carabidae of Czech and Slovak Republics. Kabinet, Zlin, p 565


41. MAYR S, WOLTERS V, DAUBER J 2007 Ground beetles (Coleoptera: Carabidae) in anthropogenic grasslands in Germany: effects of management, habitat and landscape on diversity and community composition. Wiad entomol 26: 169-184
42. MAZALOVÁ M, ŠIPOŠ J, RADA S, KAŠÁK J, ŠARAPATKA B, KURAS T 2015 Responses of grassland arthropods to various biodiversity-friendly management practices: Is there a compromise?

APPENDIX 1: LIST OF COLLECTED SPECIES (IN ALPHABETICAL ORDER)

Agonum gracilipes (Duftschmid, 1812)  
Amara aenea (De Geer, 1774)  
Amara bifrons (Gyllenhal, 1810)  
Amara communis (Panzer, 1797)  
Amara convexior Stephens, 1828  
Amara curta Dejean, 1828  
Amara familiaris (Duftschmid, 1812)  
Amara ovata (Fabricius, 1792)  
Amara plebeja (Gyllenhal, 1810)  
Amara spreta Dejean, 1831  
Amara tibialis (Paykull, 1798)  
Anisodactylus nemorivagus (Duftschmid, 1812)  
Calathus erratus (C.R. Sahlberg, 1827)  
Calathus fuscipes (Goeze, 1777)  
Calathus melanocephalus (Linné, 1758)  
Carabus granulatus Linné, 1758  
Cychrus caraboides (Linné, 1758)  
Cymindis angularis Gyllenhal, 1810  
Harpalus affinis (Schrank, 1781)  
Harpalus anxius (Duftschmid, 1812)  
Harpalus griseus (Panzer, 1796)  
Harpalus latus (Linné, 1758)  
Harpalus lateicornis (Duftschmid, 1812)  
Harpalus pumilus Sturm, 1818  
Harpalus rubripes (Duftschmid, 1812)  
Harpalus rufigalis Sturm, 1818  
Harpalus rufigipes (De Geer, 1774)  
Harpalus smaragdinus (Duftschmid, 1812)  
Harpalus tardus (Panzer, 1796)  
Panagaeus bipustulatus (Fabricius, 1775)  
Poecilus cupreus (Linné, 1758)  
Poecilus lepidus (Leske, 1785)  
Poecilus versicolor (Sturm, 1824)  
Pterostichus melanarius (Illiger, 1798)  
Pterostichus niger (Schaller, 1783)  
Pterostichus oblongopunctatus (Fabricius, 1787)  
Syntomus truncatellus (Linné, 1761)  
Synuchus vivalis (Illiger, 1798)  
Zabrus tenebroioides (Goeze, 1777)