ORIGINAL PAPER

The influence of riparian buffer strips on carabid beetle (Coleoptera, Carabidae) assemblage structure and diversity in intensively managed grassland fields

Lorna J. Cole · Ruth Morton · William Harrison · David I. McCracken · Duncan Robertson

Abstract The use of riparian buffer strips as a means of reducing diffuse pollution in European grassland systems is becoming more common and consequently there is a need for comprehensive research on the influence of buffer strip management, placement and structure on biodiversity. Carabid assemblages were examined in a range of riparian zones in intensively managed grassland in Scotland. Carabids were monitored by pitfall trapping in riparian zones open to grazing, wide riparian buffer strips (>5 m wide), narrow boundary strips (<2 m wide) and adjacent fields in order to determine factors influencing carabid diversity and assemblage structure. While carabid diversity was greater in open riparian zones and narrow boundary strips when compared to the adjacent fields, it was actually poorer in wide riparian buffers when compared to open zones thus indicating wide buffers may actually be detrimental to carabid diversity. Carabid assemblages in wide riparian buffers were, however, more distinct from the adjacent field than narrow boundary strips or riparian zones open to grazing. Consequently, while the presence of wide riparian buffers may not promote carabid diversity within the actual buffer strips, by adding an additional habitat that supports a distinct carabid assemblage, riparian buffer strips may promote diversity at the landscape level. Carabid assemblage structure was driven by a combination of soil and vegetation characteristics in addition to physical attributes including distance from the watercourse and width of the strip. Only when we have a better understanding of the factors influencing biodiversity within riparian buffer strips can we start to formulate effective management prescriptions that fuse their dual function of pollution mitigation and biodiversity promotion.

Keywords Agri-environment · Intensive grassland management · Field margins · Ground beetle diversity · Pollution mitigation

L. J. Cole (✉) · R. Morton · W. Harrison · D. I. McCracken · D. Robertson
Land Economy and Environment Research Group, Scottish Agricultural College, Auchincruive, Ayr KA6 5HW, UK
e-mail: Lorna.Cole@sac.ac.uk

Introduction

Many declining farmland species utilise agricultural field margins (e.g. flowering plants (Marshall and Moonen 2002), invertebrates such as the bumblebee (Bäckman and Tiainen 2002), birds such as the cirl bunting (Peach et al. 2001) and mammals such as the water vole (Rushton et al. 2000)) and consequently the restoration and preservation of field margins is of key importance in conserving biodiversity within the agricultural landscapes of Europe. Restoration and sympathetic management of field margins in intensively managed land has been widely advocated in Europe as a way of integrating economic and environmental objectives on farmland (Boatman 1994; Jörg 1994). While riparian buffer strips (uncultivated strips at the edge of waterways) are increasingly being used to reduce diffuse pollution in grassland systems by creating a physical barrier between the water and the source of the pollution (e.g. pesticide sprayers or livestock), their role in promoting biodiversity has largely been overlooked. With comprehensive research into the influence of these riparian buffers on biodiversity lacking, it is difficult to determine the best way to manage them for biodiversity in order to obtain multiple ecological benefits (e.g. pollution control and biodiversity conservation) within agricultural landscapes.

The sympathetic management of the outer margins of cereal crops (i.e. conservation headlands) benefits a wide range of plants, invertebrates, birds and mammals (Fuller 2000; Kromp and Steinbeger 1992; Röser 1988; Siriwardena et al. 1998), and similarly managing the outer margins of grassland fields for conservation promotes the abundance of slugs, caterpillars, homopteran bugs (Haysom et al. 2000, 2004) and phytophagous beetles (Woodcock et al. 2007). Riparian zones provide a diverse array of habitats, landforms and communities and are consequently naturally rich in biodiversity (Corbacho et al. 2003). Their sympathetic management has been shown to promote butterflies, reptiles, amphibians and birds (Marshall and Moonen 2002). Riparian buffer strips therefore not only target high-quality habitats within intensively managed grassland, but can also act as wildlife corridors linking such areas within an otherwise homogenous mosaic of intensively managed land.

The age, width, length and connectivity of buffer strips are important when considering their benefit to biodiversity. When viewed solely as a means of reducing diffuse pollution, the optimal width of the buffer depends on several factors such as slope of hill, physical attributes of the soil, stocking density, agro-chemicals applied and their mode of application. Recommended widths for enhancing biodiversity are generally much wider than those suggested for diffuse pollution control (Fischer and Fischenich 2000) and it has been suggested that the minimum width should be the width that allows the buffer strip to act as a functional ecosystem rather than simply an ecotone between two habitats (e.g. more than 5 m for invertebrates: Bedford and Usher (1994)). Kaule et al. (1983) showed that narrow field margins were almost insignificant in linking uncultivated grassland habitats and species with narrow ecological ranges remained as isolated populations furthermore several studies have found that the number of bird species is positively correlated with the width of riparian buffer strips (Deschênes et al. 2002; Hafner and Brittingham 1993).

The aim of the current study was to gain insight into the factors influencing carabid diversity and assemblage structure within riparian zones in intensively managed grassland. Such information will help to determine the optimum prescription for the placement and management of riparian buffer strips.

Methods

Study sites

A total of seven grassland dominated farms in the Cessnock catchment area, Ayrshire, Scotland (UK National Grid Reference: NS 53) were selected for study. Within these farms 22 sampling locations were established, on intensively managed grassland fields (i.e. productive ryegrass, *Lolium perenne* L., swards with high inputs of inorganic fertilisers encompassing livestock grazing and/or cutting for silage). Each sampling location was allocated into one of three categories: Open Sites (i.e. sites with no fence between the field and watercourse), Boundary Sites (i.e. sites with narrow fenced off riparian strips, width <2 m, established primarily to contain livestock/mark farm boundaries) and buffer sites (i.e. sites with wide fenced off riparian strips, width >4 m, established to increase biodiversity and/or reduce diffuse pollution). At each site, two sampling transects were established, one adjacent to the watercourse (Water) and the other 4–6 m into the field from the fenceline, or in the case of open sites 4–6 m from the Water transect (Field). For buffer sites a third transect was established, where appropriate, at the midpoint between the fenceline and the Water transect (Middle). A total of seven different treatments were therefore investigated (Fig. 1, Table 1). The study sites were chosen specifically to represent a range of vegetation types and widths of strips.

Sampling methodology

Carabids were sampled by pitfall trapping over a 3 years period (2004–2006). At each transect a line of pitfall traps was installed (total 51 lines). Each line of pitfalls consisted of 9 plastic cups, 75 mm diameter and 100 mm deep placed at 2 m intervals. Each trap contained approximately 50 ml of mono-propylene glycol (i.e. to act as a killing agent and preservative) and was covered by a 15 mm wire mesh to reduce interference by livestock and small mammals (Cole et al. 2002). Traps were installed in June and left in situ for
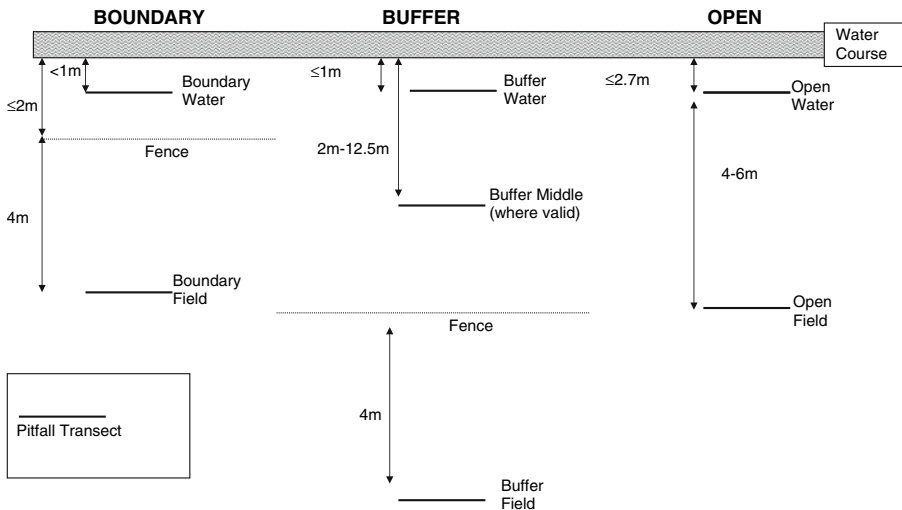
Fig. 1 Experimental setup of the survey sites indicating the three types of riparian zones (Boundary, Buffer and Open) and the two–three pitfall transects within each, thus giving a total of seven treatments (Table 1)

Table 1 Description of treatments, their codes, and the number of farms and transects in each

Treatment	Code	No. transects	No. farms	Description
Buffer Water	BufW	6	2	Adjacent to water course in riparian buffer strip
Buffer Middle	BufM	5	1	In centre of riparian buffer strip
Buffer Field	BufF	6	2	4 m into field from buffer/field fence line
Boundary Water	BW	6	3	Adjacent to water course in boundary strip
Boundary Field	BF	6	3	4 m into field from boundary/field fence line.
Open Water	OW	9	6	Adjacent to unfenced water course
Open Field	OF	9	6	4 m into field from open water line

4 weeks. Following collection, the traps were reinstalled and left for a further 4 weeks giving two annual uplifts (June and July). On collection, the samples from the nine pitfalls in each line were pooled. As both the activity and abundance of carabids influence pitfall catches (Greenslade 1964), the number of carabids caught will be referred to as the activity abundance (Thiele 1977). Carabids were identified to species level in order to determine effects on assemblage structure and diversity.

Information on vegetation height, density and composition was collected annually in autumn for each transect. Vegetation composition was determined by randomly placing four 1 m × 1 m quadrates along each pitfall line and recording percentage bare ground, plant species diversity, relative abundance of species (recorded on the domin scale) and frequency of key weed species (i.e. number of *Cirsium* and *Rumex* spp.). The Robel pole visual obscurity method (Robel et al. 1970) was used to measure sward density and height.

During pitfall installation in June four soil cores (6 cm diameter and 10 cm deep) were taken at random from each line of pitfall traps and the soil was subjected to standard soil analyses to determine: pH, % moisture content, % organic matter content, phosphorus availability (mg/l) and potassium availability (mg/l). Information on the soil impenetrability (blf/in^2) was collected using a soil penetrometer during pitfall installation and collection. Data on the transect altitude, width of margin and distance of transect from the water course were also collected.

Statistical analyses

Canonical correspondence analysis (CCA: ter Braak and Šmilauer 2002) was conducted, without downweighting rare species, to determine the principal environmental components driving the carabid assemblage structure. Prior to analysis data from the two sampling dates in each year were combined to give an overall indication of carabid assemblage structure for that year. This analysis was conducted on the carabid relative abundance data to overcome problems in estimating true densities associated with pitfall trapping.

A total of 13 continuous environmental variables and four categorical variables were considered for analysis (Table 2). A fifth categorical variable (Open Water) was omitted as a consequence of collinearity. Where required, environmental variables were log/arcsin square root transformed to normalise prior to analysis. To reduce problems with multicollinearity

Table 2 List of environmental variables taken into consideration in the canonical correspondence analysis

Factor	Description
Continuous	
Margin width*	Width of fenced off margin where pitfall transect is located (m)
Organic matter*	Soil organic matter content %
Moisture*	Soil moisture content %
Dicotyledon spp.*	Number of dicotyledon plant species
Vegetation density*	Density of vegetation as measured by Robel Pole (cm)
Impenetrability*	Soil impenetrability (blf/in^2)
Altitude*	Altitude of the pitfall trap location (m)
Distance from water*	Distance of pitfall transect from nearest water (m)
pH	Soil pH
Phosphorus	Availability of phosphorus in the soil (mg/l)
Potassium	Availability of potassium in the soil (mg/l)
Bare ground	Percentage cover of bare ground
Grass spp.	Number of grass species
Categorical variables	
Buffer Water*	Buffer Water transects
Buffer Middle*	Buffer Middle transects
Boundary Water*	Boundary Water transects
Field	All transects established in fields
Open Water	Open Water transects (omitted due to collinearity)

* Denotes factors significant at $P < 0.05$. Significant factors are listed in decreasing order of significance

between variables, a forward selection process was applied and only variables found to be statistically significant (at the 5% level) by the Monte Carlo Permutation test were included in the analysis (ter Braak and Šmilauer 2002).

To investigate the influence of treatment on diversity a combination of diversity measures were calculated: S (number of species) and Margalef's index to measure species richness, Pielou's index to measure evenness, and Shannon and the Reciprocal of Simpson's Dominance to measure overall diversity. In addition the carabid activity abundance (i.e. number trapped by pitfalls) was calculated. By combining indices which look at evenness, richness and overall diversity a more accurate picture of diversity can be obtained (Magurran 1988). Diversity indices were calculated for the second uplift only as this uplift coincided with vegetation sampling.

Influences of treatment (Table 1), vegetation density and year of sampling on carabid diversity, abundance and species richness were determined by linear mixed models using the method of residual maximum likelihood (REML: Payne et al. 2005). As the vegetation density was recorded in autumn, only the July pitfall uplifts were included in these analyses as these data corresponded more closely to the vegetation data. Where required diversity measurements were log transformed to normalise prior to analyses. Three models were fitted in GenStat 8.1 (Payne et al. 2005) to enable effects of vegetation density, year and treatment to be separated (i.e. Fixed effects = Vegetation density + Year + Treatment; Treatment + Vegetation density + Year; Year + Treatment + Vegetation density). In each model effects of treatment, vegetation density and year were assessed after adjusting for effects of farm and sampling location (i.e. Random effects = Farm/Location/Year).

Linear mixed models were also used to determine the influence of treatment on vegetation density, vegetation height, number of grass species, number of dicotyledonous species and abundance of key weed species (i.e. number of *Cirsium* and *Rumex* spp.). For these

analyses the following model was utilised: (i.e. fixed effects = Year + Treatment; random effects = Farm/Location/Year).

Results

Over the three sampling years a total of 15,939 carabids consisting of 50 species were collected by pitfall trapping.

Canonical correspondence analysis of the data yielded eigenvalues of 0.247, 0.182, 0.131 and 0.106 accounting for 6.9%, 5.1%, 3.7% and 2.9% (for axes 1–4 respectively) of the total variation in carabid assemblage structure. The resultant ordination indicated that treatment was a major factor determining the separation of sites in the ordination space with influences of treatment being greater than those of farm, location or year of sampling (Fig. 2). For most transects, the three sampling years occurred in close proximity to one another in the ordination space indicating that carabid assemblage structure remained relatively constant over the sampling period.

Pitfalls established in wide buffer strips (both Middle and Water) had higher axis one scores than those established in Fields indicating that the carabid assemblages in the buffer strips were distinct from those in the Fields. The higher axis one scores of Boundary Water transects when compared to Boundary Field transects indicated a greater similarity of the former to Buffer transects. There was a tendency for Buffer Water transects to have lower axis two scores when compared with Buffer Middle transects suggesting that within the buffer strip proximity to the water course influenced assemblage structure. While there was a clear separation between Field and Water transects in treatments closed to livestock grazing (i.e. Boundary and Buffer) there was no observable separation between the Open Field and Open Water transects.
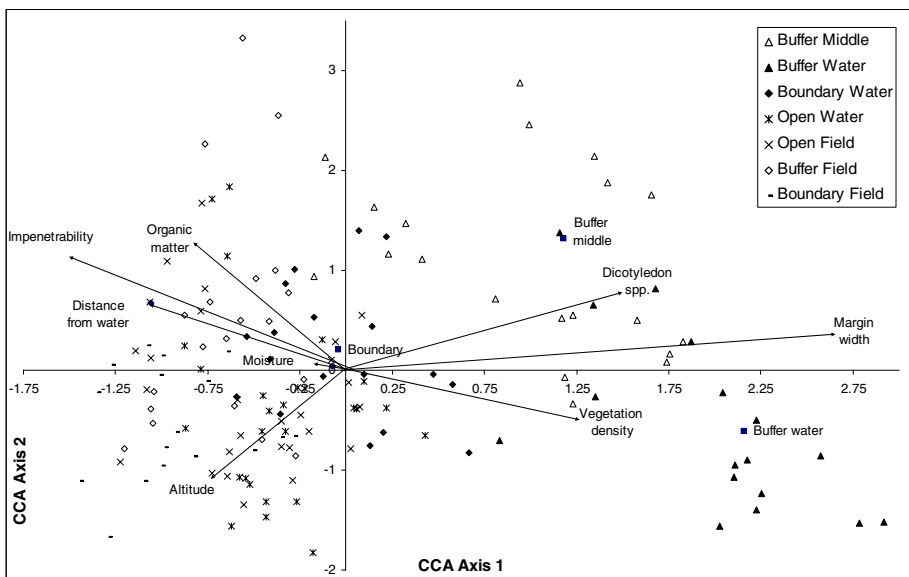
Fig. 2 Biplot derived from canonical correspondence analysis of the carabid relative abundance data highlighting the different treatments (Table 1), continuous environmental variables (vectors) and categorical environmental variables (■). Only environmental factors significant at the 5% level are included in the analysis (Table 2)

The hydrophilic, shade loving species, *Leistus rufescens* and *Stomis pumicatus*, were more frequently found in transects closed to livestock (i.e. Boundary and Buffer) than in transects where livestock had access (i.e. Open Water and all Field: Table 3). Closed transects had denser vegetation and consequently provided shadier, more humid conditions. *Agonum fuliginosum* and *Patrobus atrorufus*, again shade loving species, were also less frequently found in field transects when compared to closed transects (i.e. Boundary Water, Buffer Middle and Buffer Water: Table 3). However, these species were also frequently found in Open Water transects indicating that vegetation density was not the primary factor of influence. All Field and Open Water transects had higher frequencies of species that are typical of damp lowland grassland such as *Clivina fossor* and *Trechus micros* (Table 3). The riparian species *Bembidion schueppeli* was restricted to Open Water and Buffer Water transects, but was noticeably absent from Boundary Water transects (Table 3). *Synchus nivalis*, a seed eating species, was more abundant within Buffer (i.e. Water and Middle) and Open Water transects than Boundary Water, and Field transects (Table 3).

Eight continuous environmental variables (margin width, organic matter, moisture, dicotyledon spp., vegetation density, impenetrability, altitude and distance from water) and three categorical variables (Buffer Water, Buffer Middle and Boundary Water) influenced the carabid assemblages at a probability of <0.05% (Table 1). These eleven variables accounted for 25.7% of the observed variation in carabid assemblage structure. Soil penetrability, organic content and distance from the water course appeared to be the main environmental factors driving the division of sites to the left of the ordination indicating that transects with low axes one scores were further from the water, had a higher organic content and more compacted soil. In agreement with this, transects to the left of the ordination

Table 3 Mean percentage of transects containing dominant species (i.e. those occurring in over 50% of transects within a treatment)

	Boundary Field	Buffer Field	Open Field	Open Water	Boundary Water	Buffer Water	Buffer Middle
Bembidion schueppeli	0	0	0	22	0	57	0
Leistus rufescens	0	0	0	22	50	43	33
Stomis pumicatus	0	0	11	11	50	14	17
Synuchus nivalis	0	14	11	33	0	29	50
Patrobus atrorufus	0	29	22	67	67	100	67
Notiophilus biguttatus	17	0	11	22	50	14	17
Bembidion tetracolum	17	0	22	56	17	43	0
Agonum fuliginosum	17	14	33	44	33	71	67
Elaphrus cupreus	50	29	44	44	50	43	33
Pterostichus strenuous	50	71	89	89	100	86	100
Bembidion guttula	67	14	33	33	67	0	0
Pterostichus vernalis	67	29	67	44	67	14	50
Trechus micros	67	86	67	67	33	29	50
Pterostichus madidus	67	86	89	78	67	71	83
Amara plebeja	100	43	89	67	67	14	50
Clivina fossor	100	71	78	67	33	14	33
Bembidion aeneum	100	86	67	33	83	29	67
Agonum muelleri	100	100	100	100	100	71	83
Loricera pilicornis	100	100	100	100	100	100	83
Nebria brevicollis	100	100	100	100	100	86	83
Pterostichus melanarius	100	100	100	100	83	86	100
Pterostichus niger	100	100	100	100	83	71	100
Pterostichus nigrita agg.	100	100	100	100	100	100	100

Species are placed in order of increasing percentage in Boundary Field transects

tended to be field transects (i.e. Open Field, Boundary Field and Buffer Field). Dicotyledon species richness, margin width and vegetation density appeared to drive the separation of transects to the right of the ordination indicating that sites with high axis one scores had richer, denser vegetation and wider margins. This is in agreement with the higher axis one scores observed for Buffer Water and Buffer Middle transects.

Year did not influence any of the vegetation characteristics measured; treatment however significantly influenced all but the abundance of *Rumex* spp. (Table 4). Unsurprisingly, transects which were closed to grazing (i.e. Boundary Water, Buffer Middle and Buffer Water) had both taller and denser vegetation than transects open to grazing (i.e. all Field and Open Water transects). While Buffer Middle transects had taller, denser vegetation than Buffer Water transects, the number of both monocotydonous and dicotyledonous species was lower in Buffer Middle transects when compared to Buffer Water transects. All Water transects (i.e. Open, Boundary and Buffer) had a higher number of grass and Dicotyledonous species than the adjacent Field transects (i.e. Open, Boundary and Buffer) thus indicating the greater vegetation species richness of transects immediately adjacent to water courses.

While the abundance of *Rumex* spp. was not influenced by treatment, the abundance of *Cirsium* spp. was, with all Water transects having a higher abundance of *Cirsium* spp. than their adjacent Field transects. While the abundance of *Cirsium* spp. was greater in Buffer Water than Open Water transects, this was not the case for the Buffer Middle transects, thus indicating that the increase in the abundance of this weed within the buffer strips was restricted to the water course. Furthermore the abundance of *Cirsium* spp. in Open Field transects did not significantly differ from either Boundary Field or Buffer Field transects indicating that ungrazed riparian margins did not promote the abundance of *Cirsium* spp. in the adjacent field.

When added to the model first, treatment significantly influenced all diversity measures investigated with the exception of carabid species richness (Table 5). Four indices (i.e. Margalef, Pielou, Shannon and Simpson's) found Boundary Water transects were more diverse than Boundary Field transects. Simpson's index indicated that Open Water sites were more diverse than Field sites (i.e. Open Field, Boundary Field and Buffer Field), while Shannon's index found Open Water transects to be more diverse than any of the Buffer transects (i.e. Water, Middle and Field). Influences of treatment on Shannon and Simpson's diversity indices (but not on Margalef and Pielou's indices) were consistent following the

Table 4 Results of REML analyses to investigate the influence of Treatment and Year on vegetation characteristics

Variable	Wald Statistic (6 df); χ^2 probability	Location of difference
Grass species richness	67.65 $P < 0.001$	All Water > All Field; BufW > BufM&OW
Dicotyledon species richness	76.88 $P < 0.001$	OW&BufW > All Field&BufM; BW > BF
Rumex spp.	8.26 $P =$ NS	–
Cirsium spp.	71.53 $P < 0.001$	BufW > OW > All Field; BW > BF; BufW > BufM
Vegetation height	102.07 $P < 0.001$	BW, BufM & BufW > All Field&OW; BufW > BufM
Vegetation density	112.93 $P < 0.001$	BW, BufM & BufW > All Field&OW; BufW > BufM

As no year effect was found this information is omitted from the table. Codes for the treatments are provided in Table 1

Table 5 Results of REML analyses investigating the influence of vegetation density, treatment and year on carabid diversity and abundance

Factor	Diversity index	Factor in model first Wald & Probability	Factor in model last Wald & Probability	Location of difference
Treatment (6 df)	Species richness	7.49 P = NS	5.37 P = NS	–
	Margalef's	13.65 P < 0.05	11.70 P = NS	BW&OW > BufF&BF
	Log Pielou's	15.69 P < 0.05	11.17 P = NS	OW&BW > BF
	Shannon's	18.38 P = 0.005	18.08 P < 0.01	OW > BufW,BufM,BufF &BF; BW > BF
	Simpson's diversity	17.36 <0.01	15.16 <0.05	OW>OF,BF,BufF; BW > B
	Log activity density	56.07 P < 0.001	23.95 P < 0.001	OW,BF,OF&BufF > BW, BufW&BufM
Vegetation density (1 df)	Species richness	2.67 NS	1.18 P = NS	–
	Margalef's	2.04 P = NS	0.11 P = NS	–
	Log Pielou's	10.90 P < 0.001	6.57 P = 0.01	Gradient = 0.0042
	Shannon's	0.03 P = NS	0.10 P = NS	–
	Simpson's diversity	1.93 P = NS	0.00 P = NS	–
	Log activity density	30.96 P < 0.001	1.99 P = NS	Gradient = −0.0107
Year (2 df)	Species richness	13.22 P = 0.001	13.31 P = 0.001	2006 > 2004
	Margalef's	7.03 P < 0.05	6.19 P < 0.05	2006 > 2004
	Log Pielou's	0.34 P = NS	0.47 P = NS	–
	Shannon's	6.47 P < 0.05	6.19 P < 0.05	2006 > 2004
	Simpson's diversity	3.62 P = NS	3.35 P = NS	–
	Log activity density	17.90 P < 0.001	17.65 P < 0.001	2005&2006 > 2004

Significance of variables when included in the model first and last are provided. When significant, gradient (for vegetation height) and location of significant differences (for Year and Treatment) are provided

removal of variation as a consequence of vegetation density and consequently appear to be true treatment effects and not simply the result of changes in pitfall sampling efficiency in different vegetation densities. While carabid assemblages were therefore more diverse along watercourses open to grazing and within narrow boundary strips, the same trend was not found for wide riparian buffers and there is evidence that wide riparian buffers could actually be detrimental to carabid diversity.

Vegetation density influenced carabid activity abundance with fewer individuals being trapped in denser vegetation (Table 5). This difference, however, was only apparent when vegetation density was added to the model first, when the variation attributed to treatment was removed, the influence of vegetation density on activity abundance became insignificant. When looking at the effects of treatment, activity abundance was greater in transects open to grazing (i.e. Open Water and all Field transects) than transects closed to livestock (i.e. Boundary Water, Buffer Water and Buffer Middle transects). As treatment effects were still apparent following the removal of variation due to vegetation density, the observed differences were likely to be true treatment effects and not simply the result of a decrease in sampling efficiency in denser vegetation.

Discussion

Influence of riparian management on carabid diversity and activity density

Within the agricultural landscape riparian zones provide a diverse array of habitats, land-forms and communities and consequently are generally perceived to be rich in biodiversity

(Corbacho et al. 2003; Sala et al. 2000; Fischer and Fischenich 2000). Fencing off riparian zones to exclude livestock consequently targets an area of high biodiversity in agricultural ecosystems in addition to protecting the watercourse from agro-chemical drift and livestock contamination. In the current study the number of plant species (both monocotyledonous and dicotyledonous) was greater in grazed and ungrazed riparian margins when compared to the adjacent fields highlighting their biodiversity value. Furthermore carabid assemblages in riparian zones open to grazing and in narrow boundary strips (i.e. <2 m wide) were more diverse than the adjacent fields. Previous studies have found that wider riparian buffer strips tend to support more species of plants (Schippers and Joenje 2002) and birds (Deschênes et al. 2002). In the current study, however, there was no significant difference between the number of plant or carabid species recorded in narrow boundary strips when compared to wide riparian buffer strips (i.e. >5 m wide). Furthermore, carabid assemblages in open riparian zones were actually found to be more diverse than those in wide riparian buffer strips thus indicating that wider buffer strips may actually be detrimental to carabid diversity. Telfer et al. (2000) found that the area of bare ground within field margins had a greater impact on carabid species richness than margin width highlighting the importance of management practices and vegetation structure in determining the value of field margins to carabids. The riparian buffer strips under investigation here were fenced off and left unmanaged and as a result the vegetation was extremely dense and overgrown. Current environmental prescriptions that advocate the fencing off and hence result in the subsequent neglect of riparian buffer strips may therefore prevent them fulfilling their true biodiversity potential. It is, however, important that any management undertaken in riparian buffer strips should not unduly interfere with their role in the control of diffuse pollution. For example, while limited grazing of buffer strips could benefit biodiversity by opening up the vegetation and preventing the encroachment of woody species (Thomas et al. 2002; Olson and Wäckers 2007), by allowing livestock access to the watercourse it interferes with their role in limiting diffuse pollution. Management strategies that involve mechanically cutting or disking the vegetation, on the other hand, are labour intensive and may often be unfeasible due to the difficulty in moving machinery in the narrow strips.

Haysom et al. (2004) found that carabid densities in grassy field margins were lower than the adjacent fields and in agreement with these findings the activity density of carabids was lower in riparian strips closed to grazing when compared to the adjacent fields and open riparian zones. As pitfall catches are influenced by vegetation density, with fewer individuals being trapped in denser vegetation (Thomas et al. 1998), the lower abundance of carabids in closed riparian strips could potentially be the result of the ungrazed strips having denser vegetation. Differences in carabid density were, however, apparent following the removal of variation due to vegetation density and consequently differences between grazed and ungrazed riparian zones appeared to be true treatment effects and not simply the result of a decrease in pitfall sampling efficiency in denser vegetation. It is important to note that in the current study sampling was conducted during June and July and consequently seasonal fluctuations in the utilisation of the riparian strips by carabids may not be apparent. Grassy field margins provide an important hibernating site for many species of carabids and high densities (i.e. up to 900 individuals m^{-2}) have been found in grass margins during winter (Kromp and Steinberger 1992; Wallin 1989). The use of ungrazed riparian strips as overwintering habitats by carabids may therefore be underestimated in this study as a consequence of the restricted period of sampling.

Influence of riparian management on carabid assemblage structure

Previous studies have found that carabid assemblages in grassy field margins differ from that of the adjacent fields (Haysom et al. 2004; Kromp and Steinberger 1992). In agreement with these findings carabid assemblages of wide riparian buffer strips were distinct from intensively managed grassland fields. Furthermore, it was found wide buffer strips were more distinct from the adjacent field than either narrow boundary strips or riparian zones open to grazing. This is contrary to the findings of Telfer et al. (2000) who found that the width of conservation headlands on arable land had no effect on carabid assemblage structure. While the presence of wide riparian buffer strips may not promote carabid diversity within the buffers themselves, by adding an additional habitat, with a different carabid assemblage, riparian buffers may promote carabid diversity at the landscape level. *Leistus rufescens* and *Stomis pumicatus*, both hydrophilic shade loving species, were more frequent in ungrazed riparian strips (i.e. both narrow boundary and wide buffer strips) than in riparian zones open to grazing. Both vegetation density and dicotyledonous species richness influenced the carabid assemblage structure and it is likely that these species were favoured by the denser vegetation in the ungrazed riparian strips resulting in a more humid microhabitat. *Stomis pumicatus* specialises in eating molluscs and consequently its higher abundance could also be related to the higher abundance of molluscs that tend to be present in denser, more humid ungrazed margins (Cole et al. 2007; Haysom et al. 2004). Bohan et al. (2000) found a spatial association between the abundance of *Pterostichus melanarius* and slugs and suggested that this species aggregated in areas of high slug density. Similarly the greater occurrence of the phytophagous, *Synchus nivalis*, in the wide riparian strips and ungrazed riparian zones when compared to the adjacent fields, could be related to food availability as such sites have a higher dicotyledon species richness. In agreement with this, several studies have noted a positive correlation between the occurrence of the phytophagous genera *Amara* and *Harpalus* and weed density in cereal monocultures (Kromp 1990).

Economic implications of riparian buffer strips

Establishing riparian buffers, whether to reduce diffuse pollution or to promote biodiversity, has obvious economic implications. Land owners not only face the loss of income from taking land out of production, but also the initial costs of putting up fences and the continuing costs of maintaining them. Furthermore, it is commonly perceived that unmanaged field margins can harbour pests and weeds that may invade the adjacent field and reduce productivity (Thomas et al. 2002). In this study there was evidence that *Cirsium* spp. (but not *Rumex* spp.) were more abundant in wide riparian buffer strips than in open riparian zones. This increase, however, appeared to be restricted to riparian buffer strips and fields adjacent to buffer strips had no greater abundances of this key weed than fields adjacent to open riparian zones. Smith et al. (1999) similarly found no evidence that the abundance of economically important weeds in the field was related to their abundance in the field margins and Marshall (1989) suggested that very few field margin plant species colonise the adjacent crop. Smith et al. (1999) suggested that encouraging non-invasive grasses and wildflowers in the field margins can help to control potentially invasive weeds and it is likely that increasing the floristic diversity of the riparian buffers will also benefit a range of birds and arthropods.

Conclusion

With European research primarily focussing on non-riparian field margins (Kleijn and Verbeek 2000; Marshall 1989; Thomas et al. 2001; Woodcock 2007) or non-agricultural riparian margins (Bell et al. 1999), research on riparian margins within agricultural land in Europe is lacking. With the adoption of riparian buffer strips for diffuse pollution control becoming more prevalent there is consequently a need for comprehensive research on the influence of buffer strip management, placement and structure on biodiversity. Only once we have a better understanding of the factors influencing biodiversity within riparian buffer strips can we start to formulate effective management prescriptions that fuse their dual function of pollution mitigation and biodiversity promotion.

References

Bäckman JPC, Tiainen J (2002) Habitat quality of field margins in a Finnish farmland area for bumblebees (Hymenoptera: Bombus and Psithyrus). Agr Ecosyst Environ 89:53–68

Bedford SE, Usher MB (1994) Distribution of arthropod species across the margins of farm woodlands. Agr Ecosyst Environ 48:295–305

Bell D, Petts GE, Sadler JP (1999) The distribution of spiders in the wooded riparian zone of three rivers in western Europe. Regul River 15:141–158

Boatman ND (1994) Field margins: integrating agriculture and conservation. British Crop Protection Council Monograph no 58, British Crop Protection Council

Bohan DA, Bohan AC, Glen DM, Symondson WOC, Wiltshire CW, Hughes L (2000) Spatial dynamics of predation by carabid beetles on slugs. J Anim Ecol 69:367–379

Cole LJ, McCracken DI, Dennis P, Downie IS, Griffin AL, Foster GN, Murphy KJ, Waterhouse T (2002) Relationships between agricultural management and ecological groups of ground beetles (Coleoptera: Carabidae) on Scottish farmland. Agr Ecosyst Environ 93:232–336

Cole LJ, McCracken DI, Baker L, Parish D (2007) Grassland conservation headlands: their impact on invertebrate assemblages in intensively managed grasslands. Agr Ecosyst Environ 122:252–258

Corbacho C, Sanchez JM, Costillo E (2003) Patterns of structural complexity and human disturbance of riparian vegetation in agricultural landscapes of a Mediterranean area. Agr Ecosyst Environ 95:495–507

Deschênes M, Bélanger L, Giroux JF (2002) Use of farmland riparian strips by declining and crop damaging birds. Agr Ecosyst Environ 95:567–577

Fischer RA, Fischenich JC (2000) Design recommendations for riparian corridors and vegetated buffer strips. EMRRP Technical Notes Collection (TN EMRRP-SR-24). US Army Engineer Research and Development Center, Vicksburg

Fuller RJ (2000) Relationships between recent changes in lowland British agriculture and farmland bird populations: an overview. In: Aebischer NJ, Evans AD, Grice PV, Vickery JA (eds) Ecology and conservation of lowland farmland birds. British Ornithologist's Union, Tring, UK

Greenslade PJM (1964) Pitfall trapping as a method for studying populations of Carabidae (Coleoptera). J Anim Ecol 33:301–310

Jörg E (1994) Field margin—strip programmes. Landesanstalt für Pflanzenbau und Pflanzenschutz, Mainz

Hafner CL, Brittingham MC (1993) Evaluation of a streambank fencing program in Pennsylvania. Wildl Soc Bull 21:307–314

Haysom KA, McCracken DI, Roberts DJ, Sotherton NW (2000) Grassland conservation headlands: a new approach to enhancing biodiversity on grazing land. In: Rook AJ, Penning PD (eds) Grazing management: the principles and practice of grazing for profit and environmental gain within temperate grassland systems, 159–160 British Grassland Society Occasional Symposium 34 Harrogate, 29 February–2 March 2000

Haysom KA, McCracken DI, Foster GN, Sotherton NW (2004) Developing grassland conservation headlands: response of carabid assemblage to different cutting regimes in a silage field edge. Agr Ecosyst Environ 102:263–277

Kaule G, Beutler A, Hasase R, Scholl G, Seidl F (1983) Forschungsvorhaben: Trennwirkung von Flubereinigungswegen und Bedeutung von Rainen und Banketten. Universit Stuttgart, Abschlußbericht

Kleijn D, Verbeek M (2000) Factors affecting the species composition of arable boundary vegetation. J Appl Ecol 37:256–266

Kromp B (1990) Carabid beetles (Coleoptera, Carabidae) as bioindicators inbiological and conventional farming in Austrian potato fields. Biol Fertil Soils 9:182–187

Kromp B, KH Steinberger (1992) Grassy field margins and arthropod diversity: a case study on ground beetle and spiders in eastern Austria (Coleoptera: Carabidae; Arachnida: Aranei, Opiliones). Agr Ecosyst Environ 40:71–93

Magurran AE (1988) Ecological diversity and its measurement. Princeton University Press, Princeton

Marshall ELP (1989) Distribution patterns of plants associated with arable field margins. J App Ecol 26:247–257

Marshall EJP, Moonen AC (2002) Field margins in northern Europe: their functions and interactions with agriculture. Agr Ecosyst Environ 89:5–21

Olson DM, Wäckers FL (2007) Management of field margins to maximise multiple ecological services. J App Ecol 44:13–21

Payne RW, Harding SA, Murray DA, Soutar DM, Baird DB, Welham SJ, Kane AF, Gilmour AR, Thompson R, Webster R, Tunnicliffe Wilson G (2005) GenStat 8.1. Lawes Agricultural Trust, Rothamsted

Peach WJ, Lovett LJ, Wotton SR, Jeffs C (2001) Countryside stewardship delivers cirl buntings (Emberiza cirlus) in Devon, UK. Biol Conserv 101:361–373

Robel RJ, Briggs JN, Dayton AD, Hulbert LC (1970) Relationships between visual obstruction measurements and weight of grassland vegetation. J Range Manag 23:295–297

Röser B (1988) Saum und Kleinbiotope Ökologische Funktion, wirtschaftliche Bedeutung und Schutzwürdigkeit in. Agrarlandschaften Ecomed, Landsberg/L, pp 258

Rushton SP, Barreto GW, Cormack RM, MacDonald DW (2000) Modelling the effects of mink and habitat fragmentation on the water vole. J Appl Ecol 37:475–490

Sala OE, Chapin III FS, Armesto JJ, Berlow E, Bloomfield J, Dirzo R, Huber Sabwakd E, Huenneke LF, Jackson RB, Kinzig A, Leemans R, Lodge DM, Mooney HA, Oesterheld M, Poff NL, Sykes MT, Walker BH, Walker M, Wall DH (2000) Global biodiversity scenarios of the year 2100. Science 287:1770–1774

Schippers P, Joenje W (2002) Modelling the effect of fertiliser, mowing, disturbance and width on the biodiversity of plant communities of field boundaries. Agr Ecosyst Environ 93:351–365

Siriwardena GM, Baillie SR, Buckland ST, Fewster RM, Marchant JH, Wilson JD (1998) Trends in the abundance of farmland birds: a quantitative comparison of smoothed Common Bird Census indices. J Appl Ecol 35:24–43

Smith HE, Firbank LG, Macdonald DW (1999) Uncropped edges of arable fields managed for biodiversity do not increase weed occurrence in adjacent crops. Biol Conserv 89:107–111

ter Braak CJF, Šmilauer P (2002) CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 45). Microcomputer Power, Ithaca

Telfer MG, Meek WR, Lambdon P, Pywell RF, Sparks TH (2000) The carabids of conventional and widened field margins. Asp Appl Biol 58:411–416

Thomas CFG, Parkinson L, Marshall EJP (1998) Isolating the components ofactivity-density for the carabid beetle *Pterostichus melanarius* in farmland. Oecologia 116:103–112

Thomas CFG, Parkinson L, Griffiths GJK, Fernandez Garcia A, Marshall EJP (2001) Aggregation and temporal stability of carabid beetle distributions in field and hedgerow habitats. J Appl Ecol 38:100–109

Thomas SR, Noordhuis R, Holland JM, Goulson D (2002) Botanical diversity of beetle banks - Effects of age and comparison with conventional arable field margins in southern UK. Agr Ecosyst Environ 93:403–412

Thiele HU (1977) Carabid beetles in their environment. Springer Verlag, Berlin

Wallin H (1989) Habitat selection, reproduction and survival of two small carabid species on arable land: a comparison between *Trechus secalis* and *Bembidion lampros*. Ecography 12:193–200

Woodcock BA, Potts SG, Pilgrim E, Ramsay AJ, Tscheulin T, Parkinson A, Smith REN, Gundrey AL, Brown VK, Tallowin JR (2007) The potential of grass field margin management for enhancing beetle diversity in intensive livestock farms J Appl Ecol 44:60–69