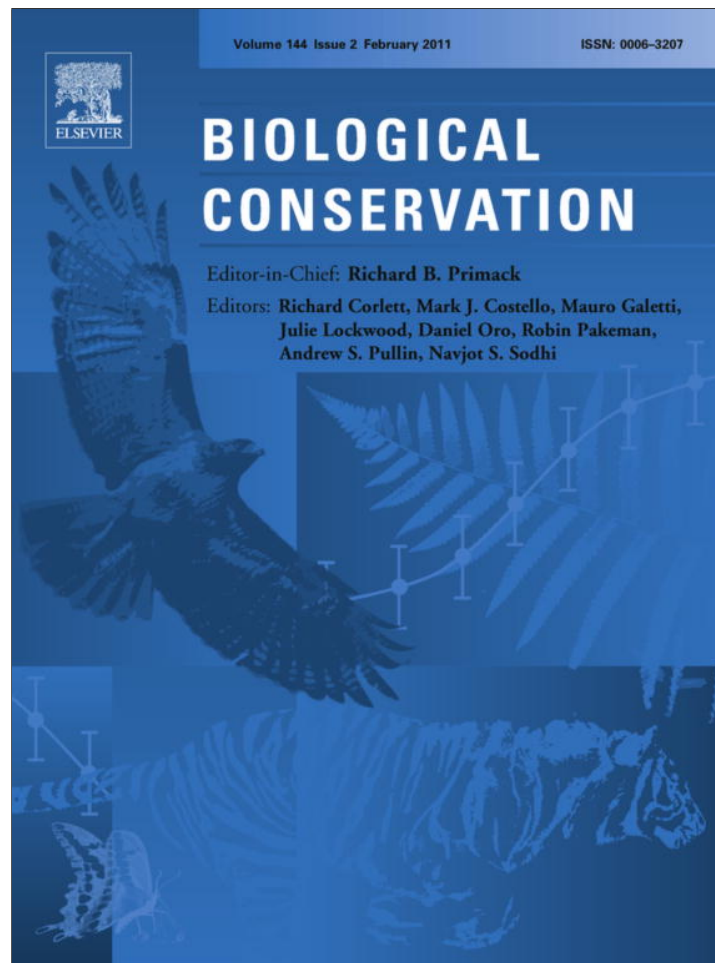


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## Review

# Effects of land management on the abundance and richness of spiders (Araneae): A meta-analysis

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## ARTICLE INFO

## Article history:

Received 26 August 2010  
 Received in revised form 17 November 2010  
 Accepted 27 November 2010  
 Available online 28 December 2010

## Keywords:

Agroecosystem  
 Edge effect  
 Forest logging  
 Grazing  
 Insecticide  
 Organic farming

## ABSTRACT

Human land use, through forest management and conversion of natural habitats into agroecosystems, has often resulted in loss of biodiversity. Spiders are important predators in terrestrial ecosystems, biological control agents against pests in agroecosystems and forests, and useful indicators due to their sensitivity to changes in habitat structure and microclimate. Effects of land use on spiders have been reviewed several times but existing reviews are narrative and usually focused on one single kind of management. We summarize the impacts of land use on spider abundance and species richness using meta-analysis. The main ecosystems where the effect of management on spiders has been studied were forests, agroecosystems and rangelands (including meadows and grasslands). Although our survey retrieved studies from all parts of the world and climatic zones, a dominance of studies from temperate habitats in Europe and North America was evident. The meta-analysis showed negative effects on spider species richness and abundance for agroecosystems and rangelands, but were less evident for forests. From 10 kinds of land management identified, all but forest plantation showed effects on spider richness or abundance. These land management scenarios affected either species richness (forest fragmentation and logging, farmland abandonment, and miscellaneous management including plowing, cutting and fire), or spider abundance (forest fragmentation, forest fire, conventional farming, insecticides, grazing, and farmland abandonment), but not both. Edge effects had contrasting effects depending on the ecosystem, affecting spider species richness in agroecosystems and abundance in rangelands and forests. Direct destruction of spiders or negative effects on habitat heterogeneity or prey populations were the most likely causes of the negative effects detected.

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## 1. Introduction

The influence of humans on ecosystems is pervasive (Foley et al., 2005; Vitousek et al., 1997). Humans use or co-opt about 24–39% of the potential terrestrial primary productivity (Haberl et al., 2007; Vitousek et al., 1986) and have transformed or degraded 39–50% of land surface (Vitousek et al., 1997). Forest destruction, management and replacement by plantations has been intensive in the last 300 years (Foley et al., 2005), first in temperate zones and more recently in tropical ones (Lewis, 2006). Croplands and pastures occupy now ca. 40% of the land surface (Foley et al., 2005; Wood et al., 2000) and have increased linearly in extent during the last 40 years (Tilman et al., 2001).

This intensive use of land and natural resources has resulted in a 10% decrease in the potential of land to yield “direct benefits” (sensu Daily, 1995). In addition, human land use or management had caused species extinctions (Pimm and Raven, 2000; Vitousek et al., 1997). First, deforestation, forest management, logging and fire implies a substantial loss of biodiversity (Bawa and Seidler, 1998; Bengtsson et al., 2000; Brook et al., 2003; Donald, 2004; Moretti et al., 2004; Niemelä, 1999; Sodhi et al., 2009). Decreases in vertebrate diversity are better documented but studies on invertebrates are also available (Bawa and Seidler, 1998; Brook et al., 2003; Liow et al., 2001; Moretti et al., 2004; Sodhi et al., 2009). Second, land transformation into agroecosystems has often entailed a simplification of the landscape which may reduce the possibility of maintaining a high number of species (Benton et al., 2003; Donald, 2004; Green et al., 2005; New, 2005; Tschardt et al., 2005). Several studies have shown negative effects of farming on mammal, bird, weed, insect and earthworm species richness (Bengtsson et al., 2005; Green et al., 2005; Hole et al., 2005; Kleijn et al., 2001; Tschardt et al., 2005). In particular, the differences between conventional and organic farming have received increasing interest (Bengtsson et al., 2005; Hole et al., 2005). Grazing has also decreased the diversity of birds, lizards, fish, tortoises and small mammals (Fleischner, 1994).

In addition to studies on taxonomic groups, concerns on decreasing diversity due to human modification of landscape have also targeted functional groups such as pollinators (Klein et al., 2007; Winfree et al., 2009). Recent research has highlighted the important role of predators in the structure of communities (Bruno and Cardinale, 2008), which adds to the traditional interest in predators as agents of biological control (Riechert, 1998). Most interest in the conservation of predators in agroecosystems has considered insects; spiders have been relatively neglected despite their importance (see, however, Riechert, 1998). Spiders are predators present in all terrestrial ecosystems (Turnbull, 1973), among the most diverse groups on Earth (Wise, 1993) and are in the highest trophic levels among the invertebrates (Main, 1987). Most spiders are generalist predators (Riechert and Luczak, 1982), hunting mainly insects (Turnbull, 1973) in all their instars (eggs and larvae to adults) (Riechert and Lockley, 1984). Spiders are considered as agents of biological control against pests in agroecosystems (Nyffeler, 1999; Riechert and Lockley, 1984) and forests (Mason et al., 1997). Their role in biological control is a community phenomenon (Riechert and Lockley, 1984) in which a high diversity of spiders is very important (Riechert et al., 1999).

Spider diversity has been a relatively neglected conservation issue despite strong indications that land use could negatively affect this group. For instance, spider assemblages are often used as indicators of management practices because their species composition and abundance are affected by changes in habitat structure (Gunnarsson, 1990; Uetz, 1991; Wise, 1993) and microclimate (Bell et al., 2001; Marc et al., 1999). Previous narrative reviews have shown decreases of spider diversity due to grazing (Bell

et al., 2001), heavy trampling (Bell et al., 2001), cutting (Bell et al., 2001), herbicide spraying (Bell et al., 2001), pesticide use (Luczak, 1979; Riechert, 1998), field improving (Bell et al., 2001) or agricultural treatments (Luczak, 1979), as well as heathland or grassland burning (Bell et al., 2001). In addition, the effects of agricultural diversification on spider abundance have been reviewed by Sunderland and Samu (2000) in relation to the potential role of spiders as biological control agents. Agroecosystems seem to be dominated by a few ‘agrobiont’ spider species (Luczak, 1979) and habitat diversification within and around the crops increases spider density (Sunderland and Samu, 2000).

Existing reviews are narrative and usually target one single kind of management. A sound management advice should be based on quantitative evidence (Sutherland et al., 2004) and meta-analysis is a powerful tool to summarize independent studies (Arnqvist and Wooster, 1995). Our main objective is to summarize the impacts of land use on spider abundance and species richness in a quantitative way, by using meta-analysis. Our specific objectives were the following: (1) To identify the kinds of management and land use potentially influencing spider species richness and abundance, (2) Do those kinds of land management increase or decrease richness and abundance of spiders? (3) Do land uses in different systems (forests, agroecosystems and rangelands) affect spiders differently? and (4) Do different kinds of management within a system (forest or agroecosystems) affect spiders in different ways?

## 2. Materials and methods

### 2.1. Literature search

We conducted a literature search of articles published since 1980. We used the following keywords in the ISI web of Knowledge database: “spider” and “management”, “spider” and “fragmentat\*”, “spider” and “insecticid\*”, “spider” and “organic farm\*”, “spider” and “fire”, “spider” and “logging”, “spider” and “graz\*”, “Araneae” and “fragmentat\*”, “Araneae” and “insectid\*”, “Araneae” and “organic farm\*”, “Araneae” and “logging”, “Araneae” and “graz\*”, “Araneae” and “fire”. We also reviewed the literature cited by the references found. A total of 173 references were retrieved (see Appendix 1).

The following information was recorded for each reference: (1) continent, (2) climate (Mediterranean, temperate, tropical, boreal and desert), (3) system (forest, agroecosystem or rangeland), (4) kind(s) of management, (5) species richness and (6) abundance or density of individuals, all species pooled (abundance, hereafter). In those references comparing more than one treatment, we selected both extremes, i.e., the less vs. most managed site or plot. In those references estimating the effects at several moments since the management was applied, we took the information of the first time after the management. In those references providing results for several seasons, we utilized the average value of the different seasons.

Some papers gave information on more than one kind of land management (i.e., organic vs. conventional crop and field edge vs. center) or system (i.e., forest and grassland). We took each land use as an independent unit of analysis, considering the *N* of each management separately, and for this reason there are more units of analysis than references.

Strictly speaking, unmanaged forests, agroecosystems and rangeland do not exist or are very rare. As in other meta-analyses (Bengtsson et al., 2005; Sodhi et al., 2009; Winfree et al., 2009), it must be understood that ‘control’ (or ‘unmanaged’) and ‘managed’ plots differed in the kind of management recorded in each original reference, but shared any other kind of management typical of that system (e.g., farming in agroecosystems).

2.2. Statistical analysis

First, a vote counting approach (Gurevitch et al., 2001) was utilized to summarize the main results. To do it we scored each unit of analysis as reporting significantly higher values in the control sites, significantly higher values in the managed sites or not significant differences.

Second, a proper meta-analysis of the data was carried out (Gurevitch and Hedges, 1993) using the program Metawin (Rosenberg et al., 2000). For each unit of analysis, the mean, standard deviation (SD) and sample size (*N*) of the variables of analysis (richness and abundance) were recorded for the managed and the control sites. The effect size was calculated by means of the Hedges' *d*:

$$d = (M_m - M_c) / S, \quad J = 1 - \frac{3}{4df - 1}$$

where  $M_m$  is the mean of the managed treatment,  $M_c$  is the mean of the control treatment, *J* is utilized to correct for the small sample size and *S* is the pooled standard deviation.

Hedges' *d* effect size allows to conduct a weighted meta-analysis (Hedges and Olkin, 1985), i.e., a meta-analysis which gives more importance to those studies with larger *N*. A mixed model of meta-analysis was chosen because it is preferable in ecological data synthesis (Gurevitch and Hedges, 1993). The bootstrap CI

(confidence interval) was utilized to test the significance of the effect size. Effect size was considered as significant when its bootstrap CI did not include 0. The heterogeneity in the size effect between units of analysis, *Q*, was tested by means of a chi-square test. We tested the heterogeneity for the global meta-analysis and for subsets of the data set categorized by *managements* or *systems*.

One problem of conducting a meta-analysis with ecological data is poor data reporting (Gurevitch and Hedges, 1999). In our case, not all studies reported SD or lacked replication. For this reason some units of analysis could not be included in the weighted meta-analysis. To take into account also these studies, an unweighted meta-analysis (Gurevitch and Hedges, 1993) was performed using the mean to create the "I<sub>r</sub>" effect size (Hedges et al., 1999):

$$I_r = \ln(M_m) - \ln(M_c)$$

In an unweighted meta-analysis, all the units of study have equivalent weight. The unweighted meta-analysis is less powerful than the weighted one, but allows to include a higher number of units of analysis.

Another potential problem in meta-analysis is the "file-drawer problem" (Rosenthal, 1979) because the probability of publication of a given study increases if it reports significant results (Arnqvist and Wooster, 1995). To estimate the file-drawer problem we used

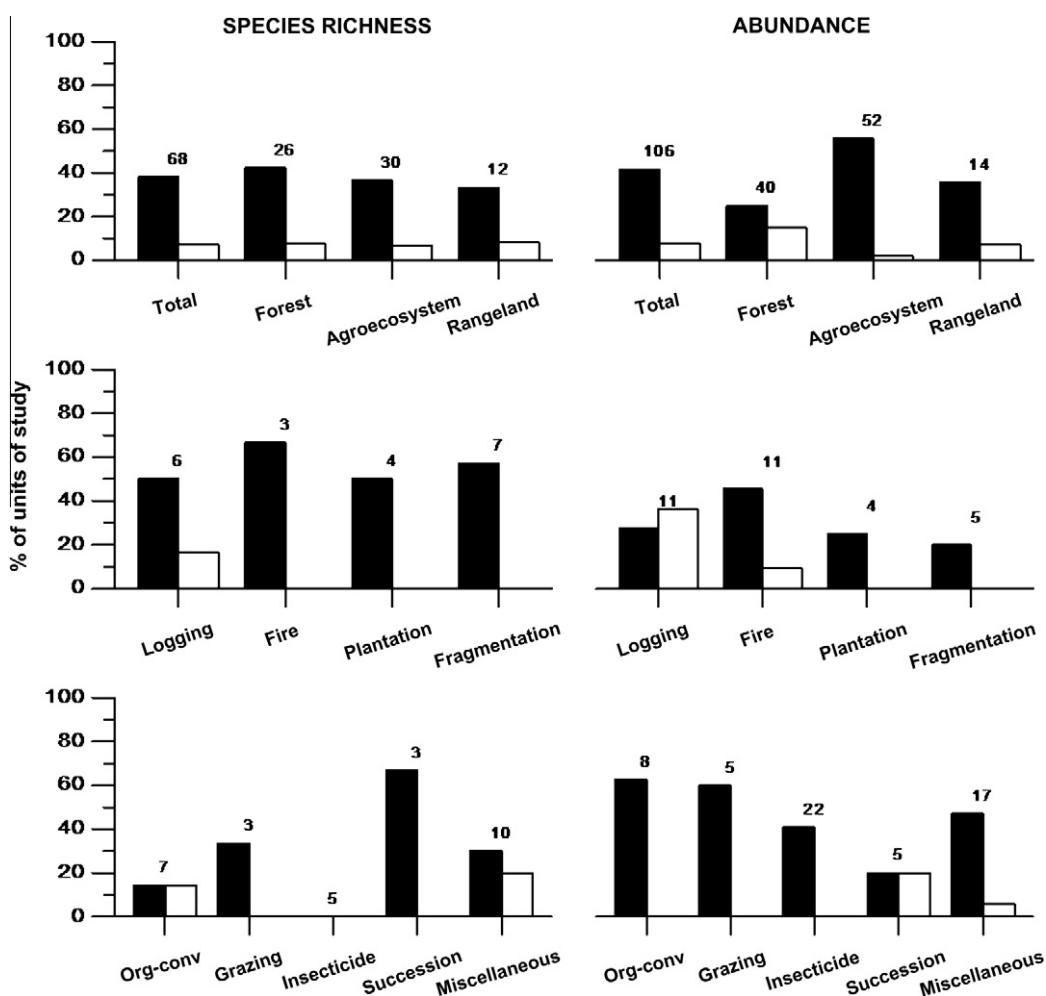


Fig. 1. Results of the vote counting for species richness (left) and abundance (right) % of units of analysis in which control sites had significantly higher values than managed sites (black bars) and in which managed sites had significantly higher values than control sites (white bars). Notice that black and white bars do not add to 100% because units of analysis reporting nonsignificant results are not pictured. Total *N* of units of analysis of each group is given above the bars.

a fail-safe number calculation. This is the number of nonsignificant, unpublished or missing studies that would need to be added to the meta-analysis to reverse the result, from significance to nonsignificance. The Rosenthal fail-safe method was utilized (Rosenthal, 1979). In addition, we utilized a funnel plot (Gurevitch et al., 2001) to visually explore the potential publication bias.

### 3. Results

#### 3.1. Kinds of management

The effect of management on spiders has been studied mainly in forests, agroecosystems and rangelands (including meadows and grasslands) (Fig. 1). Within forests, the kinds of management studied included logging, fire, fragmentation, edge effects, plantations and insecticides (Figs. 1 and 2). Within agroecosystems, the kinds of management included the comparison between organic and conventional farming, the effect of abandonment, edge effects, insecticides and miscellaneous (plowing, cutting, fire) (Figs. 1 and 2). Within rangelands, the kinds of management included grazing, succession, edge effects, insecticides and miscellaneous (plowing, cutting, fire) (Figs. 1 and 2). Most kinds of management were specific to particular ecosystems. However, edge effects had different meanings in the different ecosystems. In forests, the edge effect involved a comparison between a (usually unmanaged) forest and an adjacent open land. In agroecosystems and rangelands the edge effect involved a comparison between an unmanaged, peripheral zone and a cultivated or grazed land. Because of this, we performed a specific analysis of the edge effect and its different importance in each ecosystem.

#### 3.2. Geographic scope covered by the studies

We retrieved studies spanning the five continents and the main climatic zones of the world (Table 1). Nevertheless, over 50% corresponded to Europe and about 50% to temperate climate (Table 1).

#### 3.3. Vote counting approach

From 122 and 197 units of analysis (species richness and abundance, respectively), 42.7% did not provide any statistical analysis for species richness and 41.7% for abundance. In addition, 2.4% of the units of analysis for species richness and 5.0% of the units of analysis for abundance provided ambiguous results. These units of analysis were not included in the following description.

Overall, 38.2% of the units of analysis had more species richness in control than in managed treatments and 54.4% had similar species richness in control and managed treatments (Fig. 1). The corresponding values for spider abundance were 41.5% and 50.9% (Fig. 1). For most ecosystems and land managements considered, the most frequent result (>50% of units of study) was a lack of significant differences in species richness or spider abundance between control and managed treatments (C = M). Significantly higher species richness in control, compared to managed, sites (C > M) occurred in more than 50% of units of analysis considering logging, fire, fragmentation and succession land managements (Fig. 1). For forest plantations, C = M and C > M were equally frequent results (Fig. 1). Significantly higher spider abundance in control, compared to managed, sites occurred in more than 50% of units of analysis considering agroecosystems, organic vs. conventional farming, grazing and miscellaneous land managements

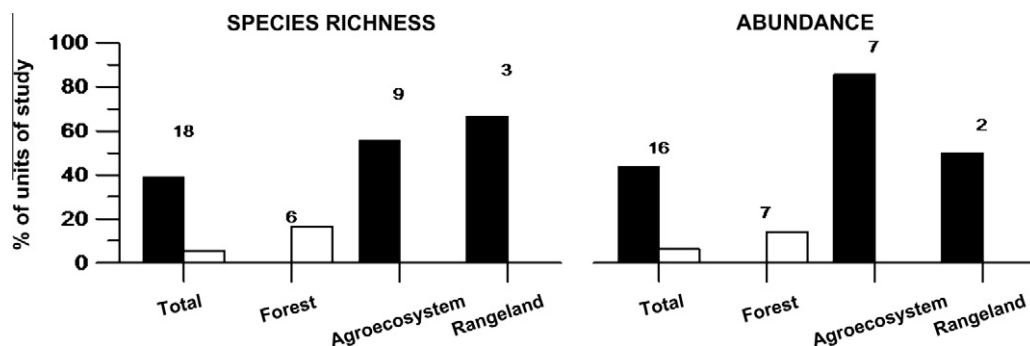


Fig. 2. Result of the edge effect vote-counting % of units of analysis in which control sites had significantly higher values than managed sites (black bars) and in which managed sites had significantly higher values than control sites (white bars). Notice that black and white bars do not add to 100% because units of analysis reporting nonsignificant results are not pictured. Total N of units of analysis of each group is given above the bars.

Table 1 Countries and climatic zones where the studies were carried out.

Continent	Climate					Total
	Boreal	Temperate	Mediterranean	Tropical	Desert	
Africa		1 (S Africa)		3 (1 Kenya, 1 Uganda, 1 Cameroon)		4
Asia	1 (Russia)	1 (Japan)		9 (2 Malaysia, 2 Taiwan, 2 India, 1 China, 1 Sumatra, 1 Pakistan)	3 (Israel)	14
Australia		1	5	6	2	14
Europe	19 (12 Finland, 5 Sweden, 1 Norway, 1 Iceland)	75 (19 UK, 16 Germany, 13 Switzerland, 5 Belgium, 5 Hungary, 5 Poland, 3 Austria, 3 Netherlands, 2 Czech Republic, 1 Denmark, 1 Bulgaria, 1 Estonia, 1 France)	6 (4 Spain, 2 Italy)	1 (Portugal)		101
N America	8 (Canada)	13 (EEUU)	9 (EEUU)	10 (9 EEUU, 1 Mexico)		40
S America		2 (1 Argentina, 1 Chile)		9 (7 Brazil, 1 Costa Rica, 1 Puerto Rico)		11
Total	28	93	20	38	5	184

(Fig. 1). For forest fire management,  $C = M$  and  $C > M$  were equally frequent results (Fig. 1).

Significantly higher species richness or spider abundance in managed, compared to control, sites ( $C < M$ ) was the less frequent result (usually  $<20\%$  of units of study) (Fig. 1). The three exceptions were: species richness in organic vs. conventional farming, where  $C > M$  and  $C < M$  were equally frequent results; species abundance in logging land management, where  $C < M$  was the most frequent result, and species abundance in succession land management, where  $C > M$  and  $C < M$  were equally frequent results (Fig. 1).

Overall, edge effects in spider richness or abundance were mostly non significant (Fig. 2). Significantly higher species richness in edges was the most frequent result ( $>50\%$  of units of study) in agroecosystems and rangelands (Fig. 2). In agroecosystems, spider abundance was higher in the edge for most units of study (Fig. 2). Less than 10% of units of study showed significantly lower species richness or spider abundance in control places (Fig. 2).

### 3.4. Meta-analysis of effects on species richness

In the weighted meta-analysis, overall data showed a significant decrease of spider species richness due to land use (Fig. 3). Land uses in agroecosystems and rangelands decreased spider species

richness (Fig. 3), and also in forests in the unweighted meta-analysis. Edges, succession and miscellaneous treatments significantly decreased spider species richness (Fig. 3). In the unweighted meta-analysis, the effects of fragmentation and logging were also significantly negative.

### 3.5. Meta-analysis of effects on abundance

In the weighted meta-analysis, overall data showed a significant decrease in spider abundance due to land use (Fig. 4). Land uses in agroecosystems and rangelands, but not in forests, significantly affected spider abundance. Fragmentation, conventional farming, grazing and insecticide significantly decreased spider abundance. Fire affected negatively spider abundance in the weighted meta-analysis but not in the unweighted one. Succession did not affect abundance in the weighted meta-analysis but it had a negative effect in the unweighted meta-analysis.

### 3.6. Partition of the edge effect

In forest ecosystems, edges had a significantly higher spider abundance, but not spider species richness, than forest interior (Fig. 5). In agroecosystems, edges had higher spider species rich-

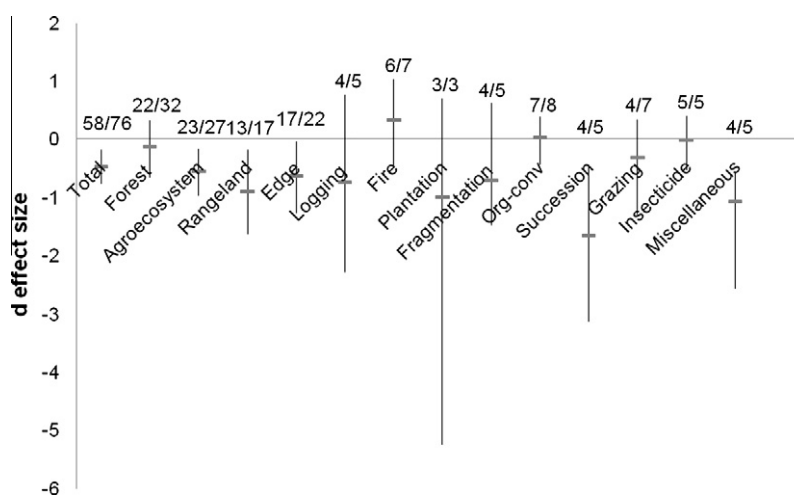


Fig. 3. Average effect size of land management on spider species richness in different ecosystems or management situations. Bars are the bootstrap confidence intervals. Sample size (no. of units of analysis) is shown above the bars as weighted/unweighted.

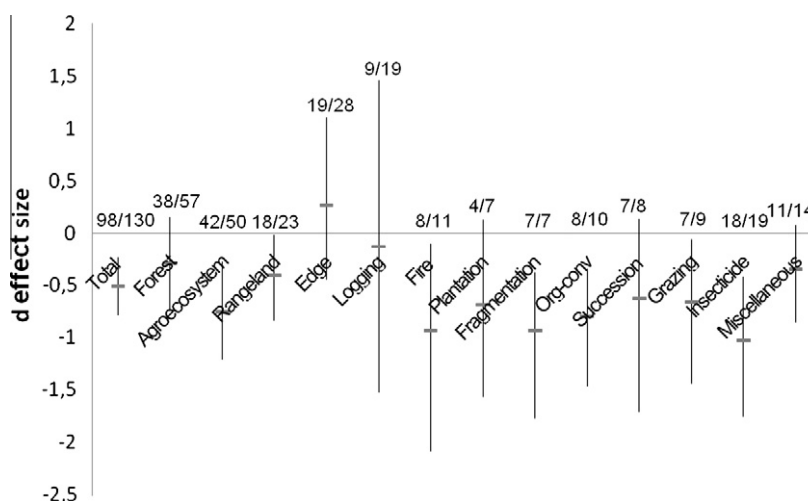
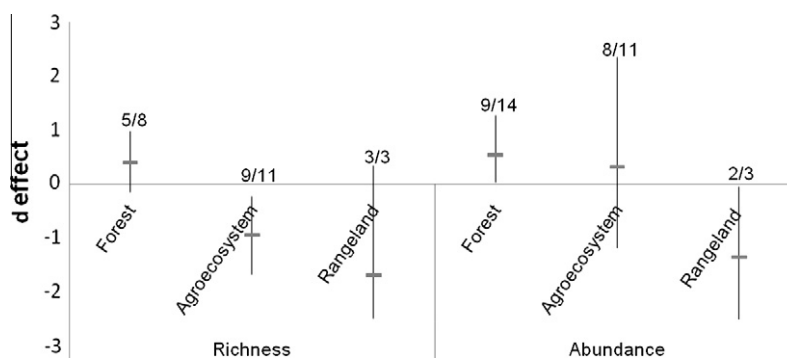


Fig. 4. Average effect size of land management on spider abundance in different ecosystems or management situations. Bars are the bootstrap confidence intervals. Sample size (no. of units of analysis) is shown above the bars as weighted/unweighted.



**Fig. 5.** Average effect size of the edge effect on spider species richness or abundance in different ecosystems. Bars are the bootstrap confidence intervals. Sample size (no. of units of analysis) is shown above the bars as weighted/unweighted.

ness, but not abundance, than managed sites (Fig. 5). In rangelands, edges had higher abundance, but not spider species richness, than managed sites only in the weighted meta-analysis.

### 3.7. Assessment of the file-drawer problem

For the weighted meta-analysis on spider species richness, 473 additional studies (fail-safe  $N$  of Rosenthal) with effect size equal 0 would be needed to reject that management had a significant effect. For the weighted meta-analysis on spider abundance the corresponding fail-safe  $N$  of Rosenthal would be of 1373 additional studies showing no effect. These fail-safe numbers were one or two orders of magnitude higher compared to the observed units of analysis, so we consider that our results are reliable for the overall effect. In the same way, the funnel plots showed a similar distribution of the units of the analysis at both sides of the 0 effect size line, indicating no publication bias (Fig. 6).

## 4. Discussion

A vote counting approach mainly indicated no significant effects of land management on spider species richness and abundance. In general, less than 40% of studies indicated significant negative effects of land management and even some studies found positive effects of management on species richness (Clough et al., 2007; Horváth et al., 2000; Jennings et al., 1988; Johnson, 1995; Tahir and Butt, 2009) or abundance (Buddle et al., 2004; Halaj et al., 2008; Jennings et al., 1988; Larrivé et al., 2005; Pearce et al., 2004).

Nevertheless, our meta-analysis showed clear negative effects of land uses in agroecosystems and rangelands on spider species richness and abundance. Negative effects of land uses in forests were less evident. The three ecosystems studied differed in the kinds, and probably the magnitude, of land management. Thus, differences between ecosystems are confounded with differences in kinds of land use implemented in each ecosystem. Nevertheless, we raise the suggestion that the apparent resilience of forest spider biota to anthropogenic disturbances can be due to the fact that these disturbances are closer in kind and magnitude to natural ones, compared to those occurring in agroecosystems and rangelands. In the following, we focus on the effects of different land uses on species richness and abundance of spiders.

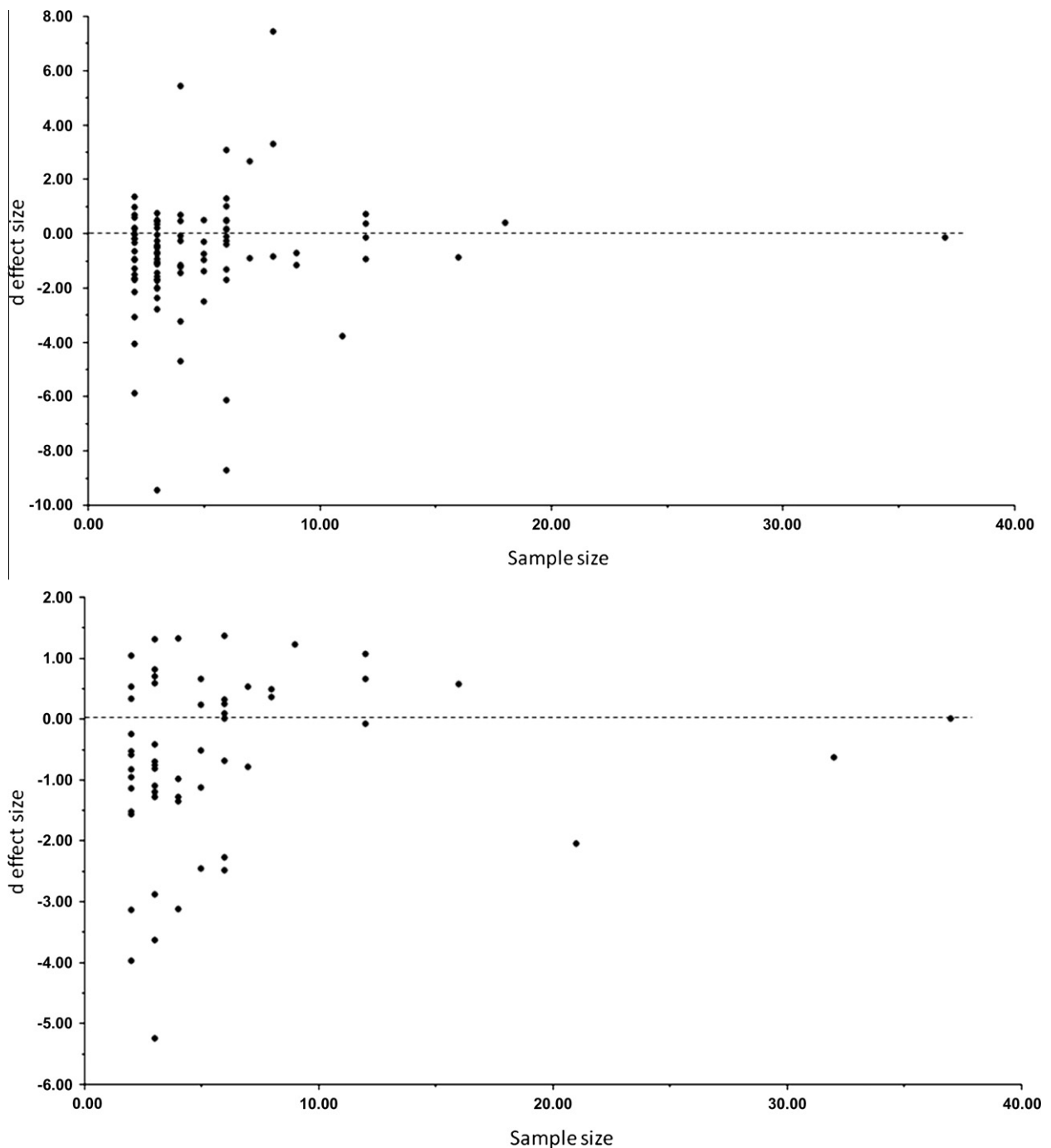
Most kinds of land management had a negative effect on spider communities. The only exception was forest plantation, which seems to offer suitable conditions for spider assemblages (Baldissera et al., 2008). The different kinds of land management influenced either species richness or abundance, but not both (at least according to the weighted meta-analysis). We are aware that species richness and abundance do not provide the whole picture

of effects of land use on spider communities. Spider species turnover from unmanaged to managed sites should also be taken into account. Its neglect in the present work can be responsible for some mismatch between the effects of land use on species richness and abundance. Species turnover was beyond the scope of the present review but will be addressed in a subsequent paper.

In agroecosystems and rangelands, management vs. succession and miscellaneous management (fires, cuttings, plowing) had a negative effect on spider species richness. This was likely related to habitat heterogeneity, which is considered important for the preservation of spider diversity and abundance in agroecosystems (Samu et al., 1999). Rangeland spiders are also very dependent on sward structure (Gibson et al., 1992). Land use in agroecosystems and rangelands included in our meta-analysis probably led to more homogeneous habitats. For example, cutting, fire (Bell et al., 2001; Thomas and Jepson, 1997) and plowing (Haskins and Shaddy, 1986; Nyffeler et al., 1994) produce changes in the vegetation structure that negatively affect spider fauna. Cutting and fires produce more extreme changes in the vegetation structure than grazing (Bell et al., 2001) and this reason could explain the differences between these managements in the effect on spider richness. In successional fields the increase of the heterogeneity due to weeds and more structured soil surface increase species richness (Sunderland and Samu, 2000; Alderweireldt, 1994).

In forests, although most kinds of management are expected to lead to higher heterogeneity due to succession, and thus to higher species richness (Niemelä, 1999), only the unweighted meta-analysis showed some significant effects of fragmentation and logging on richness. The effects of fragmentation on spider species richness were more likely related to changes in patch size and increased isolation, which reduces immigration (Miyashita et al., 1998; Floren and Deeleman-Reinhold, 2005). The effects of logging were probably related to replacement by another spider community after the big changes produced in the microhabitat (Mclver et al., 1992).

In agroecosystems and rangelands, insecticide spraying, grazing, and conventional, as opposed to organic, farming affected spider abundance negatively. Insecticides destroy spider fauna (Luczak, 1979) and reduce its abundance (Thomas and Jepson, 1997). Insecticides also seem to affect spider diversity (Pekár and Kocourek, 2004) but this effect was not shown in our analyses. Grazing affects spider abundance through direct effects on vegetation structure and microclimate by trampling and nibbling, and through indirect impacts on prey availability (Bell et al., 2001; New, 2005). As in Gibson et al. (1992), abundance was negatively affected in grazed sites, but not species richness. In the case of organic farming, our results agree with those of a previous meta-analysis (Bengtsson et al., 2005) which showed higher abundance than in conventional farming. The complexity of the agricultural landscape also seemed to increase spider richness in Agri-



**Fig. 6.** Species richness (upper panel) and abundance (lower panel) funnel plots. The horizontal dashed line shows an effect size of zero. Only the units of study included in the weighted meta-analysis are shown.

environment schemes (AES) (Concepción et al., 2008), but our meta-analysis did not reveal a significant effect on richness. The effectiveness of organic farming depends on landscape complexity, and only the right management at intermediate levels of complexity could lead to more biodiverse sites (Concepción et al., 2008; Tschardt et al., 2005).

In forests, higher abundance in larger patches could be explained because these large fragments are more resistant to edge effects and stochastic extinction (Ozanne et al., 2000), as well as higher habitat heterogeneity in larger patches (Rosenzweig, 1995). Fire greatly modified microhabitat (loss of the canopy closure and litter structure). These changes in the forest-floor charac-

teristics could be bigger after fire than after logging and communities may vary more after prescribed fires than clearcutting (Buddle et al., 2000; Larrivé et al., 2005). However, overall effects of fire affected only spider abundance and not species richness.

Edge effects had contrasting effects depending on the ecosystem. In agroecosystems, field edges had higher spider species richness while in forests, and maybe in rangelands (weighted meta-analysis), edges had higher spider abundance. Bolger et al. (2000) suggest that an increase of spider abundance in forest edges could be due to higher prey availability mediated by the increase of non-native grasses in the edge habitat. The higher species richness



in agroecosystem edges can be explained by the higher heterogeneity of the unmanaged boundaries (Bell et al., 2001; Knop et al., 2006). The ecotone effect (Sunderland and Samu, 2000) that hosts spider species from both habitats due to spillover (Knop et al., 2006) can also be important. Firstly, spider species from non agricultural habitats are present also in field edges because foraging or mate-searching behavior (Draney, 1997). Secondly, field edges may also serve as a refuge for agrobiont species arriving by spill over from the crop (Draney, 1997). In addition, edges possibly act as buffers reducing direct effects of pesticides on spiders (New, 2005).

The present meta-analysis allows some tentative implications for the conservation of spiders in agroecosystems, rangelands and managed forests. First, reduction of mechanical disturbances (ploughing, cutting and grazing) will increase spider diversity in agroecosystems and rangelands. Second, insecticides decrease the abundance of spiders, which are natural predators in agroecosystems. All these kinds of disturbances are minimized in organic farming, which had significantly higher abundance of spiders and, probably, more efficient control of prey populations. Third, the main negative effect in forest was fragmentation, which should be avoided and reversed.

As a final remark, although our survey retrieved studies from all parts of the world and climatic zones, a clear dominance of studies from Europe and North America was evident (Nyffeler, 1999; Nyffeler and Sunderland, 2003). This might seem justifiable because those are highly developed areas where land use has been intensive and continued. Notwithstanding, agroecosystems are important in India and SE Asia, and rangelands in Africa and large portions of Asia (Foley et al., 2005), where very few studies have been conducted. A stronger emphasis should be given to the study of the effects of land use on spiders, particularly in developing Asian and African countries.

## Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.biocon.2010.11.024.

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