Heteroptera ecology, biodiversity and conservation

Ecología, biodiversidad y conservación de heterópteros

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1 Introduction

1.1 The little things that run the world

> *Ecologists have a responsibility to humanity, one that we are not yet discharging adequately. It is incumbent on senior ecologists to take the lead in pressing for the needed transformation- and we pledge ourselves to that task.*
>
> Bazzaz, Ceballos, Davis, Dirzo, Ehrlich, Eisner, Levin, Lawton, Lubchenco, Matson, Mooney, Raven, Roughgarden, Sarukhan, Tilman, Vitousek, Walker, Wall, Wilson and Woodwell (1998)

> *…let me say a word on behalf of these little things that run the world.*
>
> Edward O. Wilson (1987)

There is a general consensus that global environmental change is causing a biodiversity crisis in which a wide variety of species are becoming extinct (Wilson 1992, Pimm et al. 1995, Levin 1999, Stork 2010). How the biosphere has been actually affected by this accelerated disappearance of species is a subject of intense study and debate (Duffy 2009, Cardinale et al. 2012). To tackle this subject it becomes imperative to first understand the magnitude of what has, is being, and will be lost. Therefore, the pressing question is: with how many other species are we sharing the planet? To date, the exact number remains unknown, and will, perhaps, remain unknowable. What we do know is that the number of described species falls somewhere between the range of 1.0 to 1.8 million (Stork 1988, 1993, Wilson 1992, Pimm et al. 1995), while the number of estimated species given by different authors varies widely from 3 to 100 million (May 2010, Hamilton et al. 2010, Mora et al. 2011). Regardless of the actual number, one thing is certain, the vast majority of these species are insects (Wilson 1987, Stork 1988, 2007). In fact, the latest estimate gives the number of known insect species at 1,004,898 (Adler and Foottit 2009). Arguably, the hyperdiversity of insects is one of the most durable and empirically-tested observation in the history of science (Labandeira and Sepkoski 1993, Berenbaum 2009).

Whether as individuals or aggregated into super-organisms, insects are interesting, valuable and aesthetically pleasing life forms. Inasmuch as insect diversity is very important to ecology and society, the fundamental importance of insects for humankind only comes to light when we take into account their functional role in food webs and life cycles, and their influence on agriculture, livestock and human health. Good examples of ecological functions performed by insects include: nutrient cycling, plant pollination, seed dispersal, soil structuring and fertilization, population regulation and food provisioning (Berenbaum 1996, Waldbauer 2003, Scudder 2009). All throughout the biosphere, the functions performed by insects within ecosystems translate into a vast wealth of ecosystem services that *these little things that run the world* have the potential to provide to humankind (Daily 1997, Losey and Vaughan 2006, Kremen and Chaplin-Kremer 2007, Straub et al. 2008).

1.2 Taxonomy as a fundamental discipline

> *"What's the use of us having names,"* the bug said, *"if we won't answer to them?"*
>
> *"No use to you,"* said Alice, *"but it's useful to the people that name you, I suppose. If not, why do things have names at all?!"*
>
> *"I can't say."* the bug replied. *"Further on, in the woods down there, they've got no names!"*
>
> Adapted from Lewis Carroll's "Through the Looking Glass"

> *Our generation is the first to fully comprehend the threat of the biodiversity crisis and the last with the opportunity to explore and document the species diversity of our planet.*
>
> *The grand biological challenge of our age is to create a legacy of knowledge for a planet that is soon to be biologically decimated.*
>
> Quentin Wheeler, Peter Raven and Edward Wilson (2004)

> *No one would expect every ecologist to be an expert systematist but it is fundamental if the synthesis of results is to yield order and not chaos, that every ecologist should know accurately the plants and animals with which his studies are concerned.*
>
> Edward J. Salisbury (1939)

> *Without taxonomy to give shape to the bricks, and systematics to tell us how to put them together, the house of biological science is a meaningless jumble.*
>
> Robert M. May (1990)

Each species is unique. Unique proteins are codified in their genes. Unique behaviors emerge from their aggregated cellular functioning. Species occupy unique ecological niches and network positions within food webs. This unifying pattern is highlighted by results from a large array of studies across all levels of biological organization (Werner 1992, Gollery et al. 2006, Bascompte 2009). There is an inherited complexity emerging from this hierarchical nature of biological systems, and it is precisely the ultimate goal of biological conservation to maintain this complexity (Wilson 1992, Margules and Pressey 2000, Purvis and Hector 2000). To up for this challenge the fundamental role of taxonomy must be recognized.

It is estimated that less than 10% of the species living on planet Earth have been discovered, named and classified (Wilson 2005). It is very improbable that we will be able to preserve what we do not know. This issue becomes inordinately acute with smaller organisms belonging to poor known taxonomic groups. In the words of T.R. New (1996), "Many of the world's insects species are unlikely ever to receive formal names and will become 'neofossils' in our lost heritage." Thus, it becomes imperative to bridge the gap between taxonomy and biological conservation (Wilson 2000, 2004, Golding and Timberlake 2003, Mace 2004, Samper 2004).

A broad aim of the present thesis is to highlight how this gap may be bridged by coupling biodiversity faunistic survey efforts with the monitoring of state variables relevant to ecological quantitative research. These linked faunistic and ecological studies allow us to better understand the biological diversity that we are interested in preserving, while allowing us to make well informed conservation-oriented policy and management decisions. As a first step towards this goal, we took up the task of identifying all surveyed or collected material to species level. Albeit not exclusively, this was achieved by developing region-specific diagnostic dichotomous keys. Most available keys: (1) have been developed for larger regions (ie, include many non-relevant taxa), (2) are not up-to-date (ie, may not include the most recent taxa described or species synonymies), (3) are phylogenetically coherent but impractical for identification purposes (ie, the characters they use suffer from a low degree of observability), (4) were designed exclusively for the identification of physical specimens (ie, do not take into account in-situ photographs), and (5) were written in languages that are less accessible to the present-day scientist and/or conservationist practitioner. Here, we developed 157 dichotomous keys specifically designed to address these impediments. Another important step was to use our species data to develop faunistic catalogs and dataset. These taxonomical syntheses are essential for understanding where species occur and how they are distributed. Moreover, they may be central to identify potential issues regarding the surveyed species' conservation. Finally, we explicitly incorporated into the study the use of in-situ photography and biodiversity web resources, as we believe these are essential tools that meet the challenge of expediting taxonomical research and engaging the general public in the conservation of nature.

1.3 Heteropteran bugs

I limited the taxonomical extent of the present thesis by focusing this faunistic and ecological research on the highly-diversified, trophically-diverse and economically-important group of insects known as heteropteran bugs. Formally denominated Heteroptera Latreille, 1810, heteropteran bugs or true bugs (Figure 1.1) constitute a monophyletic clade of hemimetabolous insects presenting a worldwide distribution. In the context of taxonomical hierarchy, the Heteroptera rank as a suborder of the order Hemiptera Linnaeus, 1758. Paleobiologists place the beginning of their evolutionary line in the Mesozoic (Grimaldi and Engel 2005). Since then, they have evolved into seven clades, which are given the taxonomical rank of infraorders (Stys and Kerzhner 1975, Wheeler et al. 1993, Schuh and Slater 1995): Cimicomorpha Leston, Pendergrast and Southwood, 1954; Dipsocoromorpha Miyamoto, 1961; Enicocephalomorpha Stichel, 1955; Gerromorpha Popov, 1971; Leptopodomorpha Popov, 1971; Nepomorpha Popov, 1968; and Pentatomomorpha Leston, Pendergrast and Southwood, 1954.

According to the latest review by Henry (2009) the estimated number of described heteropteran bug species is 42,347. This estimation is partially based on the regional catalogs for North America (Henry and Froeschner 1988), Australia (Cassis and Gross 1995, 2002) and the Palearctic (Aukema and Rieger 1995, 1996, 1999, 2001, 2006). One of the goals of the present work was to provide an up to date figure of the number of described heteropteran bug species in the Iberian Peninsula, which I can now confidently give as 1,453 (1,470 when the subspecies are included). Table 1.1 shows a summary of the number of families, genera and species by infraorder for

Figure 1.1 The aposematically-colored lygaeid bug *Spilostethus furcula* (Herrich-Schaeffer, 1850) photographed at the Jardinet de l'Om urban garden (Barcelona, Catalonia, Spain). Source: original.

the world, the Australian, Nearctic and Palearctic ecozones, and Iberian Peninsula bioregion.

Heteroptera monophylogeny is based on the following three morphological synapomorphies (Carver et al. 1991, Zrzavy 1992, Wheeler et al. 1993, Schuh and Slater 1995, Henry 2009, Schaefer 2009, Weirauch and Schuh 2011): elongated feeding appendages in the form of a piercing-sucking rostrum arising from the front of the head (Figure 1.2), paired scent glands present on the metathoracic pleura of adults or in the abdominal dorsum of immature stages, and four-segmented antennae with two intersegmental sclerites. An open rhabdom of the ommatidium, a character proposed by Fischer et al. (2000) may prove to be an additional synapomorphy for Heteroptera. Interestingly, the hemelytra (ie, forewings that are anteriorly sclerotized and posteriorly membranous), a well-known heteropteran feature, is presently considered to be a derived character (Wheeler et al. 1993, Weirauch and Schuh 2011).

Most heteropteran bugs are strictly phytophagous, zoophagous or hematophagous (Schuh and Slater 1995, Schaefer 2009). Others display a wide range of mix omnivorous behaviors ranging from phytozoophagy to zoophytophagy (Alomar and Widenmann 1999, Coll and Guershon 2002, Eubanks et al. 2003). Phytophagous species feed on roots, leaves, flowers, pollen, buds, seeds, fern fronds and fungi mycelia (Figure 1.3). Zoophagous species prey upon arthropods and even small vertebrates (Figure 1.4). As will be further discussed throughout this work, predatory heteropteran bugs, through their capacity to regulate pest populations, are essential for ecosystem functioning and resilience against disturbance in human-dominated habitats and landscapes. Hematophagous species feed on the blood of birds, bats and humans. Two examples of the latter are the bed bug *Cimex lectularius* Linnaeus, 1758 (Figures A2.28 and A3.3C), which has become a serious pest in many temperate regions of the world (Reinhardt and Siva-Jothy 2007), and kissing-bugs (Reduviidae: Triatominae) (Figure 1.5), which may carry the kinetoplastid parasite responsible for Chagas disease in the American continent (Ribes et al. 2008).

As many authors have previously emphasized (Miller 1971, Dolling 1991, Schuh and Slater 1995), heteropteran bugs successfully utilize a large number of different habitats. They have been documented living in association with ants, termites, spiders, and embidinids. Some present adaptations that allow them to thrive on water surfaces (Figure 1.6) or to live a true underwater aquatic existence. Some species live only in the intertidal zone, yet others venture into the open ocean. They have been recorded from all vegetation strata in all ecozones and bioregions of the world. Through this contribution we have gained evidence for their ubiquitous occurrence in human-dominated habitats and landscapes, including private gardens, public parks, urban ruderal margins, golf courses, field margins, oldfields and vineyards.

1.4 Quantitative ecology

> *It was not until I asked myself why the larger species should breed first, and then the more general question as to why there should be two and not 20 or 200 species of the genus in the pond, that ideas suitable to present to you began to emerge.*
>
> Reflections upon observing the breeding patterns of the two aquatic heteropteran bug species *Corixa punctata* and *Corixa affinis*, which he found living in a small pond just below the sanctuary of Santa Rosalia in Mount Pellegrino, Palermo, Sicily, Italy.
>
> G. Evelyn Hutchinson (1959)

Figure 1.2 A predatory assassin-bug (Family Reduviidae) photographed in a patch of open tropical savanna near the town of Katherine (Northern Territory, Australia). The ► indicates the feeeding rostrum. Source: original

Table 1.1 Summary of the known number of heteropteran bug families, genera and species by infraorder for the Australian, Nearctic and Palearctic ecozones, as well as those for the Iberian Peninsula bioregion.

Taxon	World			Australian			Nearctic			Palearctic			Iberian Peninsula		
	Families	Genera	Species	Families	Genera	Species	Families	Genera	Species	Families	Genera	Species	Families	Genera	Species
Cimicomorpha	17	2707	20,564	10	276	624	12	342	2414	12	653	4430	7	222	805
Dipsocoromorpha	5	59	337	3	15	66	3	6	10	3	8	34	2	3	3
Enicocephalomorpha	2	65	425	2	5	7	2	5	10	2	7	16	0	0	0
Gerromorpha	8	161	2120	6	49	327	5	19	104	6	39	210	5	8	27
Leptopodomorpha	4	64	381	2	7	28	2	12	71	4	20	113	3	11	27
Nepomorpha	11	140	2309	10	32	302	8	36	248	9	41	268	7	19	60
Pentatomomorpha	42	2623	16,211	34	438	1164	23	255	987	30	750	3279	22	217	548
Total	**89**	**5819**	**42,347**	**67**	**822**	**2518**	**55**	**675**	**3844**	**66**	**1518**	**8350**	**46**	**480**	**1470**

Australian, Nearctic and Palearctic figures are based on those given in Henry (2009). Iberian Peninsula figures were estimated in this work.

Figure 1.3 The herbivore *Lygaeus simulans* Deckert, 1985 (Lygaeidae) photographed in Vall de Nuria (Gerona, Catalonia, Spain). Source: original.

Since time immemorial, humans have wondered about the quantities, ranges and diversity of other living beings and about how those living beings in general interact with the surrounding environment. Eventually these wonderings evolved into the scientific endeavor known as ecology. Formally, ecology might be defined as the scientific study of the abundance, distribution and diversity of species, and of the interactions that generate these patterns across the hierarchical scales of biological organization (Haeckel 1866, Krebs 1972, 2008, Odum and Barett 2005, Begon et al. 2006). To be able to answer ecologically-related questions, ecologist must first face two universal characteristics of the living systems they are interested in studying: complexity of causation and uncertainty (Schneider 2009). In other words, as ecologist, we may be able to determine some of the multiple interacting causes driving a given observation, acknowledging that other causes, due to their stochastic nature, will remain unknown. Interestingly and importantly, at least part of this stochastically-driven uncertainty is caused by the various ways we use to observe and measure the living world.

As mentioned above, a braod aim of the present thesis is to highlight how the gap between taxonomy and biological conservation may be bridged by coupling biodiversity faunistic survey efforts with the monitoring of state variables relevant to ecological quantitative research. Here, we dived into the inherently complex and uncertain world of living systems from the platform of quantitative ecology. Hence, we attempted to establish formal relationships among the species and environmental data derived by our observations through the use of quantitative models.

Figure 1.4 The pentatomid predatory species *Picromerus bidens* (Linnaeus, 1758) feeding on a lepidopteran caterpillar. Source: Mark Johnson (Flickr).

1.4.1 *Modeling reality*

> *A good model does not attempt to reproduce every detail of the biological system; the system itself suffices for that purpose as the most detailed model of itself.*
>
> Simon A. Levin (1992)

> *To make sense of an observation, everybody needs a model …*
>
> *whether he or she knows it or not.*
>
> *It is difficult to imagine another method*
>
> *that so effectively fosters clear thinking about a*
>
> *system than the use of a model written in the language of algebra.*
>
> A Creed for Modeling. Marc Kéry (2010)

Every day, when we observe the world around us, we realize that the outcome of events are usually partially unpredictable. For example, a pregnant woman may ask herself whether she is expecting a girl or a boy, or faced with the evidence of a partially cloudy sky we ask ourselves whether is going to rain or not. These not fully predictable systems, which are ruled by uncertainty, are denominated stochastic systems (Kéry 2010). The most adequate scientific frameworks to describe and analyze data stemming from stochastic systems are probabilistic and statistical modeling (Lindley 2006, Gelman and Hill 2007, Royle and Dorazio 2008).

To develop a model is to write down, in the *language of algebra*, the abstract mathematical relationship that we think might exist among the different elements of a stochastic system. A fundamental part of modeling is to incorporate unobservable quantities, or parameters, which can then be numerically estimated. Because models are abstract constructs of the human mind, and by our own nature we observe and measure the world imperfectly, every model is bound to be wrong. The role of trained scientists is, however, to search and find those models that give a useful insight into the systems that they study (Kéry 2010).

1.4.2 Hierarchical models

A common pattern of data, across a wide spectrum of scientific disciplines, is that they are frequently observed to be organized hierarchically. Ecological data, for example, might be designed to be sampled at a series of transects within plots,

Figure 1.5 In the American continent, reduviid kissing-bugs in the genus *Triatoma* (as the one portrayed above) are potential vectors of Chagas disease. Source: Glenn Seplak (Flickr).

Figure 1.6 Species in the genus *Gerris* present adaptations that allows them live on the surface of water. The species shown above was photographed in the Aigüestortes i Estany de Sant Maurici National Park (Lérida, Catalonia, Spain). Source: original

which might be distributed within a larger bioregion. These hierarchies can also be observed when investigating the interactions among species in an ecosystem (ie, insect species group into functional assemblages that in turn are part of a larger community). Hierarchical linear models (HLMs), the focus of this section, are generalization of regression methods that describe relationships between variables within a hierarchical dataset (Gelman and Hill 2007, McMahon and Diez 2007, Royle and Dorazio 2008). They are powerful statistical tools able to cope with complex systems in which stochasticity is acting at multiple levels (Clark 2005). In HLMs at least some of the parameters to be estimated are considered random variables, and the parameters that describe the distribution of these random variables are termed hyperparameters. A more elaborate explanation of the mathematics behind HLMs is out of the reach of the present thesis. The interested readership is invited to consult the excellent accounts of HLMs given by Gelman and Hill (2007), and, in a full ecological framework, by Royle and Dorazio (2008).

Hierarchical linear models have been successfully applied to a number of ecological, diversity and conservation challenges, including research looking at association of variables across scales (Diez and Pulliam 2007), species distributions (Gelfand et al. 2005, 2006, Latimer et al. 2006, Royle et al. 2007, Kéry et al. 2010b, Pollock et al. 2012), distribution of invasive species (Latimer et al. 2009), predicting the spread of ecological processes (Wikle 2003), assessment of coextinction risk (Moir et al. 2011), effects of human-induced disturbances (Zipkin et al. 2009, Wanger et al. 2010), effects of conservation and management actions (Russell et al. 2009, Zipkin et al. 2010), and many studies focusing on the diversity, occupancy, abundance and population trends of species that are imperfectly detected (Dorazio and Royle 2005, Dorazio et al. 2006, Kéry et al. 2005, Royle and Dorazio 2006, Royle and Link 2006, Royle and Kéry 2007, Kéry and Royle 2008, Kéry et al. 2009a, 2009b, Kéry et al.

2010a, van Strien et al. 2010, Chelgren et al. 2011, Martin et al. 2011, Wintle et al. 2012).

One of the aims of the present work is to show how hierarchical linear models may be used to estimate species richness and occupancy, and to quantify the potential effects of environmental covariates on these state variables. Moreover, as will be explained in more detail in the following sections, we specifically developed our hierarchical linear models to illustrate how they may account for the extra stochasticity acting at the level of the observational process.

1.4.3 Bayesian inference

The unknown quantities of a stochastic system (i.e., the model's parameters) can be estimated using statistical methods. The best known are the frequentist and Bayesian methods. Both methods attempt to make inferences (i.e., probabilistic conclusion about the parameters) based on a model and the empirical data observed in the system being studied (Kéry 2010). There is an ongoing, and sometimes heated, debate on which of these inference methods is most fit to analyze biodiversity and ecological models. The description of this debate is, however, out of the contextual area of this work, and I shall not discuss it further. Nonetheless, after a brief introduction to the Bayesian mode of inference, I will attempt to explain the main characteristics of the Bayesian approach to statistics, and its advantages, always in the context of ecology, diversity and conservation, over frequentist methods. In the present research Bayesian methods were exclusively used in the parameter estimation of all models. The goal is to demonstrate the use of Bayesian methods for solving biodiversity and ecological challenges.

Central to the development of Bayesian methods is the idea that a parameter's posterior distribution $p(\theta|x)$, that is, the probability distribution of obtaining the parameter θ given the data x can be calculated through the following equation:

$$p(\theta|x) = \frac{p(x|\theta) \cdot p(\theta)}{p(x)}$$

The above expression is known as the Bayes' theorem (Bayes and Price 1763), where: (1) $p(x|\theta)$ is the Fisher's likelihood function (Fisher 1922), or the information about parameter θ contained in the data; (2) $p(\theta)$ the prior probability distribution of the parameter θ, or what is known about the parameter before collecting data x; and (3) $p(x)$ a normalizing constant, or the marginal probability density that makes the integral of $p(\theta|x)$ equal to 1 (Ellison 1996, 2004, Kéry 2010).

Therefore, Bayesian methods recognize and combine different components of knowledge (McCarthy 2007). A statistical model is used to combine prior knowledge with new collected data to generate new (posterior) knowledge. Many researchers agree that this is exactly how we and other species update knowledge in our mind (see Kruschke 2010 and reference therein). I shall try to shed light on this matter with an example from my own experience. Poisonous snakes are known to occur in the areas where I collected insects for this work (the Iberian Peninsula bioregion and the Melbourne Metropolitan Area). When asked whether they had encountered snakes in the field, fellow Iberian researchers invariably replied negatively. Posed with the same question, Australian researchers generally answered that at least once or twice they had seen snakes during their field campaigns. Thus, regardless of my prior knowledge that there are indeed poisonous snakes in the Iberian Peninsula, new data stemming from fellow field researchers made me feel at ease, confident that the probability of encountering a poisonous snake was very low. On the other hand, my prior knowledge that Australian snake species where both abundant and deadly, combined with new data from fellow researchers made me very cautious and snake-aware when sampling in Australia. Luckily, in neither the Iberian Peninsula or Melbourne, have I encountered a poisonous snake when sampling for insects!

What are the benefits of Bayesian methods? One of their greatest advantages is that they allow for the estimation of the probability of a hypothesis being true. In a frequentist null hypothesis testing, a p-value is not the probability of a hypothesis being true, A p-value only represents the probability of obtaining the data given the hypothesis. Hence, frequentist methods have a strong tendency to focus on statistical significance, which can lead to irrelevant dichotomous decision making (Fidler et al. 2006), especially in management decisions regarding the conservation of biodiversity. On the contrary, Bayesian methods focus on estimating effects sizes and providing a measure of the precision of those estimates. The latter is the practice that is encouraged by ecological societies and the journals they publish (Fidler et al. 2006, McCarthy 2007). In the Bayesian mode of inference, credible intervals (CI) constructed around the parameters' means are used to represent their precision (i.e., associated uncertainty), and these CI are used to determine the ecological importance of the effect size of these parameters. Another advantage of CI is that their width provides information on statistical power (McCarthy 2007). For example, in a hierarchical analysis with a parameter that is estimated across two different scales, the wider of the two CIs will inform us the scale of the study to which we should allocate more field sampling, if we wish to achieve a similar precision for both scales. Yet, the greatest benefit of the Bayesian mode of inference is that it allows for the introduction of external knowledge into the analyses. In this respect, frequentist methods assume a complete ignorance of the system under study. With Bayesian methods ignorance must be also specified, and this is achieved using flat non-informative priors. When a Bayesian analysis is conducted with non-informative priors, the inference is based on the observed data alone, and the results will tend to be numerically similar to those obtained under a frequentist analysis (Kéry 2010).

In conclusion, inference about the unknown quantities of a stochastic system might be estimated either by frequentist or Bayesian statistical methods. As long as certain circumstances are met, both methods might yield similar results. Here, nonetheless, the goal is to demonstrate the use of Bayesian methods for inferring the parameters specified into our hierarchical linear models. My hope is that this demonstrations may shed some light into the benefits of the Bayesian mode of inference, especially those critical to ecological, diversity and conservation issues.

1.4.4 Species richness and occupancy

We saw above that one way to address the issue of the complexity and uncertainty inherent to ecological systems was to use statistical models to establish formal relationships among the measured data derived by observations (Schneider 2009). A statistical model, paraphrasing Kéry (2010), explains the variation in an observed response as being composed of a deterministic and a stochastic part. The deterministic component attempts to explain the multiple interacting causes driving the given observation, by, in the case of linear models, assuming that the expected response varies according to the combined additive effects of a series of explanatory variables (Kéry 2010). On the other hand, to specify the model's stochastic component we use probability distributions, which in turn are described by unobservable quantities. As previously discussed, these unobservable quantities or parameters are estimated using inference methods (Royle and Dorazio 2008, Kéry 2010). We are left with the third and final component of a model: the response. The variation in the response is what we actually observe, measure and attempt to explain. As in other dynamic systems in nature, within an ecological system of interest, measured responses that characterize the system status can be best understood as state variables (Yoccoz et al. 2001). By gathering, sampling, collecting, surveying, monitoring or otherwise assessing by any other observational mean, data about one or more state variable(s) within an ecological system of interest, we set the foundations of the quantitative processes that will lead us to determine some of the multiple interacting causes driving our observation, always within the margin of uncertainty associated with the ecological process driving the observations, and with the uncertainty derived from the own observation process that gathered the data.

Although ecology is concerned with the whole hierarchy of biological organization, we will focus our efforts on the species and community levels of ecological systems. Since we are interested in answering questions relating to the distribution and diversity of species, our state variables of interest will be occupancy and species richness. Occupancy can be defined as the number or proportion of spatial units in which a given species lives (Kéry and Schmidt 2008). It has been recognized as useful state variable in studies involving rare species, metapopulations and geographic distributions (MacKenzie et al. 2005). Our second state variable of interest is species richness. As the count of the number of species living in a given area, species richness is the simplest measure of biological diversity (MacArthur 1965, Adams 2009). Species richness is a state variable of special interest in studies concerned with the conservation of whole communities (Margules and Pressey 2000, Purvis and Hector 2000, MacKenzie et al. 2006).

As previously discussed, at least part of the uncertainty that we must take into account when studying ecological systems arises by the own methods we use to observe and measure the living world. Despite the importance of accounting for this uncertainty inherent to the processes of observation and measurement, many studies, including some well-funded conservation-oriented monitoring programmes, frequently overlook two key sources of error: spatial variation and detectability (Yoccoz et al. 2001). These two sources of uncertainty are discussed below.

1.4.5 *Spatial variation*

A well-recognized source of error when estimating occupancy and species richness arises as a consequence of our inability to measure these state variables over the entire area we would like to study (Yoccoz et al. 2001, MacKenzie et al. 2005, Kéry and Schmidt 2008). We are unable, metaphorically speaking, to see the whole picture. To circumvent this problem, ecologist conduct their observations of the biological system in a sample of smaller spatial units. These samples are then used to draw inferences for the larger area of interest. Because these inferences need to be as unbiased and accurate as possible, special attention should be paid to introducing a certain degree of randomness or stratification in the sample selection process (Thompson 2002). An aim of the present work is to account for the error introduced by spatial variation. As illustrated in the case studies that shall be presented below, our field methodology specifically included one or more sources of randomness and stratification in the selection process of our spatial units.

1.4.6 *Detectability*

> *Surprisingly…virtually all approaches have neglected one important aspect of ecological data – one which every naturalist knows well – almost any species may be overlooked.*
>
> Marc Kéry (2011)

As discussed above, the first source of uncertainty that we must deal with when working with count data is spatial variation. The second source of uncertainty is detectability (Boulinier et al. 1998, Yoccoz et al. 2001, Wintle et al. 2004, MacKenzie et al. 2002, 2005, Dorazio et al. 2006, MacKenzie 2006, MacKenzie et al. 2006, Kéry and Schmidt 2008, Royle and Dorazio 2008). Detectability may be best understood as a probability. When the detection probability of given count is 1, we are looking at perfect detection. In other words, the process by which we observed a given natural system was carried out without error: wherever we counted was the true number of things available for counting. In animal surveys, however, this degree of perfection is seldom, or more likely never, reached (Boulinier et al. 1998, Kéry and Schmidt 2008). Moreover, there is evidence that even immobile organisms such as plants are also not perfectly detected (Kéry et al. 2006). On the other hand, when the detection probability of a given count is less than 1, we are looking at the much more frequent phenomenon of imperfect detection.

Allow me to illustrate the issues surrounding detectability with an example. Suppose we are members of a research team tasked with documenting the distribution of an hypothetical 'bug' species, which I shall call *Detectalia overlooka* Mata 2013, in two remote islands, Frequentkey and Rarerkey, which have not ever been surveyed for insects before. We go into great lengths to design a robust and well-balanced sampling protocol: same number of plots and temporal replicates, same-sized plots and same-length surveys, and so forth. Importantly, surveys will be conducted using two collecting methods (A and B). We decide the team should split-up, Team F will go to Frequentkey, and team R to Rarerkey. When we arrive to the islands we realize that the equipment has been mixed-up, team F has brought along only equipment for collecting method A, while team R are stuck with only equipment for collecting method B. We decide to go ahead with the survey anyway. After a few weeks, the surveys are completed, and the data is explored. The F and R teams reports *D. overlooka* as occurring in 16 and 32% of Frequentkey and Rarerkey, respectively. So, apparently the species is more widely distributed in Rarerkey than in Frequentkey. Before these results are published (fortunately!), an article comes out reporting precisely on the detection efficiencies of methods A and B. As it turns out, the probability of detecting *D. overlooka* with method A is estimated at 0.2, whereas method B detects the species with a much higher probability of 0.8. How does this influence our findings? Re-exploring them in the light of the new detection data we see that actually *D. overlooka* occurs in 80% of Frequentkey, while only occurring in 40% of Rarerkey. Thus, by overlooking the issue of detectability we arrived at unrealistic conclusions regarding the actual distribution patterns of *Detectalia overlooka*. More importantly, this example highlights another pressing matter, even if methods A and B had not been mixed, without detection knowledge, we would still have underestimated occupancy.

In the example above, the uncertainty associated with the imperfect detection of the hypothetical bug species *Detectalia overlooka* was driven by our choice of sampling methodology. Imperfect detection however, is also induced by other factors, which, as described by Bouliner et al. (1998), may include: (1) differences in abundance of individual species (eg, species present in larger quantities may be detected more frequently), (2) differences in behavior (eg, a species may hide in response to the observer's presence or be more active given certain climatic conditions), and (3) differences in learning rates of the observer(s) or observation method(s) (eg, an observer may learn with time to detect more efficiently a given species or the accuracy of a method may become less precise with time). Certainly, other factors are also at play. A given plot's vegetation structure or even the morphology of plant species within a plot, may increase the odds that the organism under study may be overlooked (see for example Chapter 20 in Kéry 2010).

Given the importance of accounting for detectability in ecological studies using count data, it is surprising how frequently the issue of detectability goes 'undetected' by researchers. Studies that overlook the extra uncertainty introduced by the observation process may (Gu and Swihart 2004, MacKenzie et al. 2006, Royle and Dorazio 2008, Kéry 2010, 2011): (1) underestimate occupancy, as in our example of *D. overlooka* above; (2) underestimate the effect of covariates, which may, for example, in conservation studies, leave valuable habitats outside the scope of management actions; and (3) misidentify the effect of covariates on the observation process as being drivers of the biological process under study, as, for example, might happen if an entomologist trying to understand the distribution pattern of a given nocturnal species sets light-traps under the protective cover of a thick-branched tree, and then (wrongly) inferred that tree cover was the cause driving occupancy.

An aim of the present work is to account for the uncertainty introduced by detectability. Whenever possible, our sampling protocols were specifically designed to include spatial and/or temporal replicates from which detection data could be deduced. As reflected in the case studies presented below, we analyzed our data through statistical models that were precisely developed to include the observation process.

1.4.7 *A challenge of scale*

> *Acts in what Hutchinson has called the ‘ecological theatre’*[1] *are played out on various scales of space and time.*
>
> *To understand the drama, we must view in on the appropriate scale.*
>
> John A. Weins (1989)

> *What was the thread, if any, that had guided my wanderings?*
>
> *In retrospect, it became clear that a fascination with scale had underlain of all these efforts; it is, I will argue, the fundamental conceptual problem in ecology, if not in all of science.*
>
> Simon A. Levin (1992)

Scale can be defined as the resolution within the extent of a measured spatial or temporal quantity (Urban 2005, Schneider 2009). A study looking at insect diversity, for example, may have a scale resolved to the area of a 20 by 30 m plot, within the extent of 1,000 km^2 study region. The same study may be resolved to daily insect surveys over the extent of a week. Scaling issues are fundamental to both pure and applied ecological investigations (Weins 1989, Levin 1992). The most pressing issues with which scale has been challenging ecologists include the recognition that (Mandelbrot 1977, Weins 1989, Holling 1992, Levin 1992, 2000, Schneider 2009): (1) ecological and biodiversity patterns depend on the scale of analysis; (2) there might not be a single ‘correct’ scale at which to analyze ecological systems; (3) scaling laws might lead to fractal dimensions; (4) species respond to changes in the surrounding environment at a range of temporal scales; (5) biological diversity patterns arise through processes that are short and local in scale and are stabilized by processes that are longer and broader in scale; (6) effects at one scale might propagate to other scales; and, most importantly, (7) problems caused by global environmental change arise precisely by this propagation of effects across scales, which means that finding solutions to them might be best tackled by cross-scale research.

The four case studies presented in this thesis were all challenged in one way or another by issues of scale. Although I am aware that many investigators have offered quantitative methods to account for scale in analysis and inference (Hooten et al. 2003, Borcard et al. 2004, Keitt and Urban 2005, Diez and Pulliam 2007, McMahon and Diez 2007), the formal modeling of scale was not undertaken in the present work. One important reason was that in most cases I was certain the data lacked enough information to account explicitly for both scale and detectability. At the ‘plot level’, for example, species data were generally dedicated to generate ‘site level’ detection histories; consequently, no replicate plot-data were left to draw inferences at the ‘plot level’ (see *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*). In other cases, specially those involving site within regions (eg, *The effect of landscape functional heterogeneity on vineyard biodiversity*), there were enough plot and site data to account for detectability and scale, respectively, but not enough replicate study regions to incorporate a new ‘regional level’ module into the models. Nevertheless, I still deemed it important to take up the challenge of scale. In fact, the study cases were designed to cover at least three different ‘scales’ of increasing spatial resolution and extent, which might be considered, at least conceptually, to be hierarchically-nested. The first scale, that for convenience I shall call the 'landscape scale' was considered in the case studies: *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* and *The effect of landscape functional heterogeneity on vineyard biodiversity*. In them, attempts were made to draw conclusions for landscape-wide areas using data resolved to the area of patches of the specific land-use of interest (eg, urban green spaces, vineyards). The second scale, that I shall call the 'shire scale', was considered in *Effects of urbanization on occupancy and species richness*. Here inferences were drawn from a whole region (formally denominated a 'shire'), constituted by an aggregation of geographically and historically linked group of landscapes, using data resolved to the area of a series of urban landscapes, each one with its own urbanization legacy. The third and last scale, that I shall call the 'bioregion scale', was considered in *Estimation of species and family detectability along macroecological gradients*. In this case study, an attempt was made to draw inferences and make predictions for a whole bioregion, specifically the Iberian Peninsula, from data resolved to the area of its constituting administrative units (ie, districts and provinces). As can be noted, as we move from one scale to the other, the spatial resolution of the latter is at least as large or larger than the spatial extent of the former. Thus, fulfilling one of the goals of the present research, I was able to explore and compare how the ecological and biodiversity patterns under study varied with scale.

1.5 Case studies

The following four case studies constitute the quantitative backbone of the present thesis. They present original investigations conducted by the author, and fellow collaborators, between the years 2010-2013. Addressing a series of research questions pertinent to the scientific study of ecology, biodiversity and conservation, these case studies: (1) are grounded on a strong taxonomical foundation, (2) have heteropteran bug insects as model organisms, (3) quantify the stochastic systems under observation while facing the complexity of multiple causation and uncertainty, (4) attempt to model the living systems under study by means of hierarchical linear models, (5) demonstrate the use of the Bayesian mode of inference, (6) focus on the species and community levels of biological organization, (7) have occupancy and/or species richness as state variables, (8) take into account the observation process uncertainty derived by spatial variability and detectability, and (9) draw inferences from studies with spatial scales that varied both in resolution and extent.

1.5.1 The insect biodiversity benefits of novel grassland ecosystems in urban green spaces

Research that lead to this case study was made possible by the opportunity given to the author to participate in the project 'Ecosystems services from large urban green spaces - the biodiversity and carbon benefit of urban golf courses', an Australian Research Council Linkage Project led by Stephen Livesley, in collaboration with Amy Hahs, Caragh Threlfall, Nicholas Williams and Nigel Stork. One of the most interesting aspects of this project is that it is investigating the link between golf course structure and management and insect biodiversity. Through it, we will have an opportunity to learn more about how the management decisions related to golf course vegetation structure and vegetation elements may influence the biodiversity values within golf courses. This knowledge can then be used to guide management decisions that promote higher biodiversity values within golf courses.

Essential to the development of this case study was the observation that novel grassland ecosystems were present amongst most of the golf courses under study. A characteristic of novel ecosystems in general is that they develop as a consequence of human action. One important mechanism that leads to their genesis is the abandonment of intensively managed ecosystems (Hobbs et al. 2006). In urban golf courses, cessation of a combination of water, fertilization and vegetation management regimes over the ‘rough’ surrounding the fairways may drive these intensively managed areas into a secondary succession leading to novel grasslands. This process may be considered analog to the process leading to ‘oldfields’, in which this type of novel ecosystem develops as a consequence of the cessation of agriculture on croplands (Odum 1960). A second important characteristic of novel ecosystems is the ‘novelty’ of their biotas, and the potential of the new combination of species within them to modify ecosystem functioning (Hobs et al. 2006).

As previously stated, novel grassland ecosystems were a common vegetation feature of the studied urban golf courses. Another common feature were woodland ecosystems, which are, with a few exceptions, ubiquitous in golf courses worldwide. These intensively managed ecosystems differ from novel grassland considerably, first by being composed principally by trees and shrubs instead of herbs and grasses, secondly, and most importantly, by depending on human intervention for their maintenance.

Our main goal in this case study is to investigate how novel ecosystems may play a role in promoting higher biodiversity values within golf courses. To understand this role we assessed the community response of heteropteran bugs to managed woodland and novel grassland in urban golf courses in south-east Melbourne, Victoria, Australia. We were also interested in investigating the possible links between golf course vegetation structure complexity and biodiversity. To quantify this response we estimated the effect of vegetation density on the species-specific probabilities of occurrence of heteropteran bugs, as well as the effect on the whole community. Finally, we were concerned with the extra uncertainty introduced by the observation process in ecological studies using count data. To circumvent this issue we developed field sampling protocols and used hierarchical models that specifically accounted for spatial variation and detectability.

We ask the following research questions:

1. Do novel grassland ecosystems within golf courses contribute to higher values of heteropteran bug species richness?

2. Is this contribution different when we look at the herbivore and predatory guilds separately?

3. How is occupancy of the whole heteropteran bug community influenced by vegetation density?

4. What is the effect of vegetation density on the heteropteran bug species-specific probabilities of occurrence?

5. How is the occupancy of heteropteran bug species predicted to vary along the vegetation density gradient?

6. Is the probability of detecting heteropteran bugs similar in woodland and grassland?

1.5.2 *The effect of landscape functional heterogeneity on vineyard biodiversity*

The establishment of the European Union LIFE+ 2009 project 'Demonstrating biodiversity in viticulture landscapes' (BioDiVine) has given the author a unique opportunity to investigate in the context of vineyard ecosystems the effect of functional landscape heterogeneity on insect species richness and occupancy. Coordinated by Joël Rochard at the Institut Français de la Vigne et du Vin, the project aims at reinforcing landscape structures in vineyards to favor biodiversity restoration (Biodivine 2013). For the development of this case study we analyzed occupancy data from some 150 heteropteran bug species derived from a standardized mammal, bird and arthropod survey implemented by the BioDiVine project in a series of vineyard sites in the Penedès wine-region of the Iberian Peninsula (Goula et al. 2013, Torrentò et al. 2013). These vineyard sites were embedded in two very distinct sub-regions: (1) Castellet i La Gornal, composed mostly of simplified landscapes containing very few vegetation elements, and (2) Avinyó Nou, composed of complex landscapes containing a mix of Mediterranean oak forest, shrubland, meadows, cropland, oldfields, orchards and gardens.

One of the main research questions asked by the BioDiVine project in the Penedès wine-region was: how does animal biodiversity vary between these two contrasting sub-regions? This question is of an important ecological and conservation relevance, as landscape compositional and configurational heterogeneity has been well documented to influence animal biodiversity (Atauri and Lucio 2001, Tscharntke et al. 2005, Benett et al. 2006, Fahrig et al. 2011), including the abundance, occupancy and species richness of beneficial predatory and parasitoid insects (Thies and Tscharntke 1999, With et al. 2002, Bianchi et al. 2006, Tscharntke et al. 2007, Hendrickx et al. 2007, Gardiner et al. 2009, Thomson and Hoffmann 2009, 2010, Maisonhaute et al. 2010, Chaplin-Kramer et al. 2011). Here, we attempted to address this question by developing models that allowed for the estimation of the effect of these two sub-regions on the species richness of herbivorous and predatory heteropteran bug species.

The BioDiVine project was also interested in mapping the landscape heterogeneity in which their study vineyards were embedded. To achieve this goal they used pre-existing land-cover maps to generate circular habitat maps around the centre point of each site. These data were used in the present case study to quantify the effect of landscape heterogeneity on the herbivorous and predaceous heteropteran bug community and species-specific occurrence probabilities. Prior to our analyses, however, we re-defined the habitat maps to explicitly consider the functional relevance of each habitat to the resource requirements of heteropteran bugs. Thus, we adopted the 'functional landscape heterogeneity' framework proposed by Fahrig et al. (2011), in which the landscape elements are defined based on differences in resource requirements of species or species groups. Accordingly, we identified three classes of functional habitats for heteropteran bugs: (1) 'dangerous', providing no resource benefit plus an active cost for the insects by going into them (eg, impervious surface); (2) 'neutral', providing no resource benefit plus a potential passive cost for going into them (eg, bare soil); and (3) 'beneficial', providing one or more resources. From this latter class, we further distinguished between 'natural habitats' (eg, Oak forest, shrubland) and 'production habitats' (eg, annual crop fields, vineyards). In a natural habitat (Fahrig et al. 2011), humans are not the main consumers of the habitat's net primary production, there is an evolutionary or long-term association between the habitat and the main species living in it, and there is a low frequency and intensity of human disturbance, especially when compared to that of a production habitat. In the present case study, we summarized area data of natural habitats to develop a measure of functional landscape heterogeneity. This measure was incorporated into our hierarchical models to test it as a predictor of heteropteran bug community and species-specific occupancy. Results stemming from this analysis could be used to guide policy decisions aimed at promoting higher biodiversity values in viticulture landscapes.

Finally, the implementation of the BioDiVine project's insect surveying protocol provided an excellent opportunity to explore and compare how its two constituting methods, the flight interception and pitfall traps, may be influencing insect detectability. We used the Penedès data to quantify the effect of sampling trap type (flight interception or pitfall trap) on heteropteran bug guild and species-specific detection probabilities. Our findings may help identify which of these methods in viticulture landscape is more efficient for collecting heteropteran bugs and may provide insights into how insect detection in general is influenced by sampling methodology.

Our objectives in this case study are threefold. Our first objective is to explore and compare the insect biodiversity of the two Penedès' sub-regions under study. These sub-regions as noted previously contrasted importantly in their landscape compositional heterogeneity. To quantify this response, we estimated the effect of these two sub-regions on the species richness of herbivorous and predatory heteropteran bug species. A second objective is to understand how landscape functional heterogeneity may influence insect biodiversity in viticulture landscapes. To quantify this response we incorporated into our hierarchical models a measure of functional heterogeneity (proportion of natural habitat) as a predictor of heteropteran bug community and species-specific occupancy, and estimated the magnitude of its effect on occurrence probabilities. Finally, we are interested in investigating the efficiency of the study's insect sampling methods. To quantify this efficiency, we estimated the effects of flight interception and pitfall traps on heteropteran bug guild and species-specific detection probabilities.

We ask the following research questions:

1. How does the species richness of herbivorous and predatory heteropteran bugs compares between Castellet i La Gornal, a sub-region characterized by its simplified landscapes, and Avinyó Nou, a sub-region characterized by its heterogeneous landscapes?

2. What is the effect of proportion of natural habitat, a measure of landscape functional heterogeneity, on the species-specific probabilities of occurrence of herbivorous and predatory heteropteran bugs?

3. How is the occupancy of heteropteran bug herbivorous and predatory species predicted to vary along the proportion of natural habitat gradient?

4. What is the effect of flight interception and pitfall traps on heteropteran bug guild and species-specific detection probabilities?

1.5.3 *Effects of urbanization on occupancy and species richness*

The present case study is explicitly linked with the faunistic study *Heteroptera from el Maresme*, which shall be presented below. These two coupled studies aim at being an example of how species data derived from monitoring or survey efforts may be used simultaneously to address faunistics and ecological research questions. The spatial extent and resolution of this case study varied considerably form the other two case studies presented above. Until now an attempt has been made to draw conclusions for landscape-wide areas using data resolved to the area of patches of a specific land-use of interest (eg, golf course, vineyard). Here, by contrast, inferences are drawn for a whole region, namely 'El Maresme' shire (north-east Iberian Peninsula), using data resolved to the area of a series of urban landscapes, namely the urban area surrounding the capital city of each one of the shire's 30 municipalities. Hence, the extent of the previous studies becomes the resolution of the present one. My interest, one that extends beyond the scope of this particular case study, is to explore whether biodiversity patterns, including community, guild and species-specific occurrence and detection probabilities, vary across the different spatial scales under study.

Our main goal in the present case study is to explore how urbanization influences insect biodiversity. Urbanization is a human-driven process that transforms native ecosystems into urban ecosystems (McIntyre and Rango 2009, Gaston 2010). These urban ecosystems are characterized by their high-levels of human habitation and energy consumption, and an intensive and extensive transformation of the landscape (McDonnell and Pickett 1990, Pickett et al. 2011). But how does urbanization affect insect biodiversity? At the community level, results from a wide array of studies indicate that moderate to high levels of urbanization correlate with low levels of insect species richness (McIntyre 2000, McKinney 2008, Luck and Smallbone 2010). On the other hand, insect species-specific responses to urbanization are less well-known (McIntyre 2000, McIntyre and Rango 2009). Some species seem to strive in highly urbanized areas. This is the case of synanthropic and urbanophile species, often denominated 'urban exploiters' or 'urban taxa' (McIntyre 2000, McKinney 2002), that depend strongly on human resources to survive. These species are generally not native to the urban areas they inhabit, instead their occupancy patterns have arisen though colonization processes that have translated them from one highly urbanized area into another (McKinney 2002, 2006). Therefore, they can be considered an important example of urbanization-driven biotic homogenization (McKinney 2010). Other species show the opposite pattern. Known as 'urban avoiders' or 'rural taxa' (McIntyre 2000, McKinney 2002), these species only show high occupancy levels at low to intermediate levels of urbanization, tending to be absent from highly urbanized areas. A third group of species, that I shall call 'urban neutral', tend to be indifferent to the degree of urban disturbance of the areas they inhabit (McIntyre 2000). Thus, they are equally likely to occur at low, intermediate or high levels of urbanization.

Here, we used hierarchical linear models to estimate the effect that urbanization has on the species richness and species-specific occurrence probabilities of heteropteran bugs living in herbaceous ruderal vegetation within the largest city of each one of the 30 municipalities that constitute the El Maresme shire (north-east Iberian Peninsula). By quantifying the level of urban disturbance of each one of our inference points we were effectively generating a continuous urbanization gradient (McDonnell and Pickett 1990, McDonnell and Hahs 2008) for the whole study area. To guarantee the repeatability and comparability of our study, we needed to construct this gradient using a standard broad measure of urbanization (McDonnell and Hahs 2008). We chose to use the Weeks' index of urbanization (Weeks et al. 2005), a tested measure that efficiently integrates physical and social components of the urban landscape (Hahs and McDonnell 2006) using existing and easily available landscape and demographic data.

We ask the following research questions:

1. How does species richness of heteropteran bugs vary in El Maresme shire along its urbanization gradient?

2. Does the species richness of herbivores and predators follow the same pattern observed for the whole community?

3. What is the effect of urbanization on the heteropteran bug community?

4. What is the effect of urbanization on heteropteran bug species-specific probabilities of occurrence?

5. How is the occupancy of heteropteran bug species predicted to vary along the urbanization gradient?.

1.5.4 *Estimation of species and family detectability along macroecological gradients*

In this fourth and last case study I am interested in exploring occupancy and species distribution patterns at the regional scale. Specifically, inferences are drawn and predictions are made for a whole bioregion (Iberian Peninsula), from data resolved to the area of its constituting administrative units (ie, districts and provinces). The attempts to understand ecological patterns at this larger spatial scale, and to apply the results of the analyses to problems relating to insect conservation, places this case study in the realms of 'macroecology' (MacArthur 1972, Brown 1999, Gaston and Blackburn 2000) and 'conservation biogeography' (Whittaker et al. 2005, Diniz-Filho et al. 2010, Richardson and Whittaker 2010). The large spatial scale of this case study made impracticable, for both economic and logistic reasons, the implementation of a research-specific systematic insect survey. This disadvantage is a common feature of most macroecological investigations (Gaston and Blackburn 2000). Moreover, our knowledge of the precise geographical ranges that species occupy at larger spatial scales is, in general, poorly understood and often inadequate, a problem that is known as the 'Wallace shortfall' (Whittaker et al. 2005, Richardson and Whittaker 2010). For insect species, this shortfall of distributional knowledge may be especially challenging (Diniz-Filho et al. 2010). Here, I address these issues by explicitly linking the macroecological research presented here with the faunistic study *Catalog of the Heteroptera from the Iberian Peninsula*, which will be presented below. The development of this 'Catalog' provided distributional data for 1,470 species and subspecies comprising the Iberian Peninsula bioregion heteropterofauna. As previously stated, coupled taxonomical-quantitative studies are a good example of how data-gathering efforts can be simultaneously used to answer faunistic and ecological questions to the benefit of both pure and conservation science.

As previously discussed, an aim of the present thesis is to incorporate into our research the uncertainty introduced by detectability. In the other three case studies introduced above we specifically designed the sampling protocols to include spatial and/or temporal replicates from which detection data could be deduced and modeled. Here, however, the present-only nature of the distributional data was not conductive to account for the imperfect detection of species. This is no trivial issue, as any attempt to model species distributions without accounting for imperfect detection will lead to results in which occupancy and detection are cofounded (Kéry 2010). This impediment becomes specially acute in macroecological studies where data is generally compiled from checklists, catalogs and museum collections (Whittaker et al. 2005, Hortal 2008, Kéry et al. 2010, Beck et al. 2012).

Accepting the limitations imposed by the imperfectly-detected nature of our data, I focused the macroecological research presented here in the stochastic surveying process generating our large scale observations. Hence, I considered the distributional data provided by the *Catalog of the Heteroptera from the Iberian Peninsula* as detection/non-detection data rather than presence/absence data. The aim of this case study is to investigate the observation process stochasticity driving large scale patterns of detection, and to assess how detectability may be influenced by macroecological gradients. I began by evaluating the ease and difficulty by which heteropteran bug species have been detected in the Iberian Peninsula bioregion. To quantify these patterns, I estimated heteropteran bug species-specific probabilities of detection using hierarchical linear models. Because I was also interested in the detection patterns of heteropteran bug families, I specified into the model family-level hyperparameters. Hence, I was able to estimate the probabilities of detection of each heteropteran bug family. Furthermore, these family-level hyperparameters were themselves governed by global hyperparameters, thus it was possible to estimate the detection probability of the whole Iberian Peninsula heteropterofauna. By estimating the ease and difficulty by which heteropteran bug species and families were detected we shall be able to observe how detectability may be related to species or family traits such as body size and coloration. Moreover, these estimates shall allows us to observe how detection probabilities of well-established native species compare to that of recently-established invading taxa. Next, I evaluated how heteropteran bug detection varied along macroecological gradients. Macroecological predictors (eg, latitude and elevation) have been recognized to influence species richness, abundance and occupancy (Gaston and Blackburn 2000, Luck 2007, Diniz-Filho et al. 2010, Price et al. 2011). How these gradients may be related to species or higher-taxa detection patterns is, however, poorly understood. To address this issue, I estimated the effect of area, altitudinal range, mean annual temperature, mean annual precipitation and population density on the detection probability of the Iberian Peninsula heteropteran bug fauna, as well as on the detection probabilities of each heteropteran bug family.

I ask the following research questions:

1. What is the probability of detecting a heteropteran bug in the Iberian Peninsula bioregion?

2. What are the probabilities of detecting individual heteropteran bug species and

families?

3. Which heteropteran bug species traits relate to high detection probabilities?

4. How do detection probabilities of well-established local species compares to that of recently-established invading taxa?

5. What is the effect of macroecological gradients (area, altitudinal range, mean annual temperature, mean annual precipitation and population density) on the detection probability of the whole heteropteran bug fauna?

6. What are the effects of these macroecological gradients on each individual heteropteran bug family?

7. How is detection of the whole heteropterofauna and of each individual family predicted to vary along the macroecological gradients under study?

2 Objectives

2.1 Broad objectives

To conduct a general investigation into the ecology, biodiversity and conservation of heteropteran bugs; a highly-diversified, trophically-diverse and economically-important group of insects.

To highlight how the gap between taxonomy and conservation may be bridged by coupling biodiversity faunistic survey efforts with the monitoring of state variables relevant to ecological quantitative research.

2.2 Taxonomical objectives

To emphasize the value of elaborating new diagnostic dichotomous keys for species' identification.

To recognize the importance of local and regional faunistic catalogs and datasets for understanding the occurrence, distribution and potential conservation issues of biodiversity.

To increase awareness of in-situ photography and biodiversity web resources, which are essential tools to meet the challenge of expediting taxonomical research and engaging the general public in the conservation of nature.

2.3 Quantitative objectives

To show how hierarchical linear models may be used to estimate species richness and occupancy, and to quantify the potential effects of environmental covariates on these state variables.

To demonstrate the use of Bayesian methods for inferring the parameters specified into hierarchical linear models.

To incorporate into the quantitative analyses the uncertainty associated with spatial variation and imperfect detection, which are fundamental issues in modeling state variables such as species richness and occupancy.

To explore and compare how the ecological and biodiversity patterns under study varied with scale, which may assist in solving conservation problems that are best tackled by cross-scale research.

3 Material and Methods

3.1 Collecting heteropteran bugs in the field

3.1.1 Entomological nets

The large majority of terrestrial samples were collected using entomological nets, either by sweep-netting (Figure 3.1) or beating. Sweep-netting was used for bugs occurring on the foliage of herbaceous plants, whereas for those that occur on woody vegetation beating was the preferred method. Nets varied in their bag diameter from 20 to 40 cm. Likewise, the handle had a length that varied from 75 to 100 cm. Through sweep-netting and beating, it was possible to collect the less active heteropteran bugs found deep within the vegetation structure, and to record, at the same time, the plant or habitat specificity of the species sampled.

Stream-dwelling heteropteran bugs were sampled using a coarse-mesh aquatic net 25 cm in diameter using two different methods: (1) collecting the drifting debris after disturbing the marginal ground and aquatic vegetation of shallow creeks, or (2) directly sweeping the surface of rock pools.

Since specimens that fall in nets can be carefully released again to their host plant or habitat after having been recorded, the sweep-netting and beating methods are optimally suited for conservation-oriented surveys.

3.1.2 Aspirators

Bug specimens were removed from the entomological net using aspirators. Two kinds of aspirators were used: pooters and hand-held vacuums. When using pooter aspirators the collector must breathe air through a tube to suck in the specimen through another tube connected to a plastic container. Hand-held vacuums are small portable battery-powered devices that when turned on generate a current of air that suck specimens into a plastic head. Specimens must then be transferred from the head into the appropriate container. Hand-held vacuums proved to be of great use in *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* case study, where the field collectors had to simultaneously gather insect material of a large variety of taxa. Sometimes, bugs were searched for by getting on hands and knees and looking under rocks or among leaf litter; in these occasions the pooter aspirator proved essential.

Figure 3.1 The author sweept-netting an oldrough patch within a south-eastern Melbourne golf course. Source: Caragh Threlfall

3.1.3 *Berlese funnels*

On occasions, heteropteran bugs living in the leaf-litter and/or soil's top strata were sampled using Berlese funnels. Soil and leaf-litter samples collected in the field were placed in unlit funnels, and, after a standardized number of days, the extracted specimens were collected in a plastic container. This method was sometimes the only effective way to observe and collect small ground-dwelling heteropteran bugs.

3.1.4 *Pitfall traps*

Pitfall traps are ideally suited to collect ground-dwelling heteropteran species. Generally, a series of plastic containers are buried a pre-determined number of cm into the ground so that their top is at same level as the ground (Figure 3.2). Each pitfall trap is covered with a lid kept a few cm from the mound of the buried container, which is partially filled with a preserving liquid (e.g., a soapy solution).

3.1.5 *Flight intercept traps*

In *The effect of landscape functional heterogeneity on vineyard biodiversity* case study, insects were surveyed (in combination with pitfall traps) using a series of flight intercept traps (Figure 3.2). Two inter-crossed transparent plastic sheets were fitted above a yellow funnel, which was set at a height of approximately 1 m from the ground. The idea was for flying insects, actively attracted by the yellow funnel, to hit the transparent pans, and fall, through the funnel, into a vial containing a soapy saline solution. This solution was designed to preserve the specimens, which were collected on a weekly basis. This method was well suited to collect a large diversity of insect groups. We note, however, that the method was notably bias towards active flying taxa, and that it did not allow host plants or habitat specificity to be recorded.

3.2 Preservation methods and specialized techniques

3.2.1 *Dry mounting and killing bottles*

Historically, dry mounting has been the most traditional method of preserving most heteropteran bug taxa. It involves gluing specimens to small rectangular or triangular cardboards with a water-soluble adhesive. Before dry mounting procedures, specimens collected in the field are preserved inside killing bottles, which are containers that have been partially filled with a soft material (e.g., cotton or shredded cork) impregnated with an insect-killing chemical agent (e.g., ethyl acetate). These agents have the property of keeping the specimen flexible, which is ideal for the mounting process. Most of these chemical agents, however, have been documented dangerous to humans and even as carcinogenous. Therefore, their use was expressly avoided during the present work. As described below, we decided on the alternative method of preserving all specimens in ethyl alcohol.

3.2.2 *Ethyl alcohol*

After field collection by the different methods described in the previous sections, specimens were preserved in ethyl alcohol (70%). This method proved to be a safe and efficient preservation technique. It is worth mentioning, nonetheless, that some taxa shed their appendages when preserved in ethyl alcohol, yet others, whose coloration was derived from plant pigments, lost their natural coloration. Most species of the family Miridae are an example of the former, and the vivid-green pentatomid genus *Cuspicona* of the latter. Here, we assumed these loses by taking the extra care of keeping track of shed appendages during specimen processing and documenting natural coloration as part of the field sampling protocol.

3.2.3 *Dissection and mounting of genitalia*

In some cases, identification of a bug specimen to species level required the observation of the male or female genitalia. In this work we used the following genitalia-observation protocol. The genital segment was removed, submerged in a carved slide partially filled with lactophenol, and left at room temperature for at least 24h. After this time period the muscle tissue had generally disintegrated. The whole genital segment, or a specific genital structure removed from the segment (e.g., a paramere), was either placed back in ethyl alcohol for observation under the binocular-microscope or permanently slide-mounted in glycerine for observation under a high-magnification optical microscope.

3.3 Photographic equipment and methods

I used a single-lens reflex digital camera model K20D (Pentax Ricoh Imaging Company) fitted with a 100 mm macro lens (Cosina Co., Ltd). Speed, aperture, ISO, and focus were manually set for each photograph. After transferring the images to a computer environment, they were loaded for editing into the graphics program Photoshop CS (Adobe Systems). As I was interested in using the in-situ photographs as photographic field records, I associated each photograph to a series of field data, including at least date, locality and habitat type (Goula et al. 2012). Photographs and their associated metadata were uploaded to the image hosting and online photographic community website Flickr (Yahoo! Inc.) for storing, showing and sharing. I was also interested in uploading the heteropteran images to the global biodiversity web resource Encyclopedia of Life (see *Photographic biodiversity web resources* below). To allow their automated algorithm to bring our Flickr-stored photographs into their website's environment, I first gave each photograph a machine tag (e.g., "taxonomy:binomial=Beosus maritimus") and an Attribution-NonCommercial-ShareAlike Creative Commons license. The latter, besides being a requisite, allowed me to make the whole collection available to the public for education, scientific, and all other non-commercial purposes. I then proceeded to upload the photographs into the Encyclopedia of Life Flickr group. The full heteropteran bug photographic collection can be accessed following this link: http://www.flickr.com/photos/dingilingi/sets/72157625349132660/.

3.4 Biodiversity web resources

3.4.1 *The Encyclopedia of Life*

The goal of this web resource is to provide free global access to knowledge about life on planet Earth (Encyclopedia of Life 2013). They are committed to gather, generate and share this biodiversity knowledge in an open and trusted format. In the

Figure 3.2 The insect surveying protocol of the EU LIFE+ 2009 project 'Demonstrating biodiversity in viticulture landscapes' is shown above. The sampling system combined a flight intercept (right) and a pitfall (▶) trap. Source: Biodivine project

present work, we linked with this resource in more than one way. First, in order to gather data for some of our working examples (see for example *Pyrrhocoridae from the Iberian Peninsula*), we explored their biodiversity database in search for photographic records. Secondly, the author became an active collaborator in their project by becoming one of their 'full curators', which allowed him to review the project's organisms-related data objects. Lastly, as described in *Photographic equipment and methods* above, I expanded their biodiversity database by contributing the heteropteran bug image collection.

3.4.2 Biodiversidad Virtual

The citizen's platform Biodiversidad Virtual aims at gathering, managing, and communicating Iberian Peninsula biodiversity data through geo-referenced digital photography (Biodiversidad Virtual 2013). Their database grows by direct contributions from their members, which for each photograph submitted are requested to provide metadata (e.g., locality and date of capture) and to place it in the appropriate taxa folder. Prior to online publication, photographs are reviewed by a board of taxa-specific experts. In order to incorporate the photographic records contained in their database, we directly requested the director of the project J.M. Sesma to provide the heteropteran bug photographic metadata. Although I did not contributed, at this time, the heteropteran photographic collection to their platform, I envision that this could be done in the near future.

3.4.3 Flickr groups: Heteroptera from Australia and Heteroptera from the Iberian Peninsula

Flickr is an image hosting and online photographic community website in which approximately 51 million users share and embed more than 6 billion photographs. Photographic records belonging to the Australia and Iberian Peninsula bioregions were uploaded into the Flickr groups *Heteroptera from Australia* and *Heteroptera from the Iberian Peninsula* (Figure 3.3), respectively. The groups themselves were created by the author. As the idea was to encourage fellow Heteroptera enthusiasts and/or researchers to submit their material to the group, the Flickr photographic database was periodically searched using the appropriate keywords (e.g., Heteroptera + Iberian Peninsula). Once an eligible record had been found (ie, those associated to metadata), the photograph's author was notified and an invitation to submit the record to the group was issued. In almost all cases, photographs were added to the pool with some kind of species identification data. To guarantee that photographed species were correctly identified, I thoroughly curated each photograph following the same criteria used for physical specimens. Whenever the correct assignment of a taxa could not be guaranteed by the characters observable from the photo, it was not added to the group's dataset. For example, I believe that dorsal-view photographs of species in the genus *Carpocoris* (Figure 3.4) cannot be confidently assigned to any of the four Iberian species, because this identification requires observation of very specific characters only observable in the male genitalia. Thus, photos of *Carpocoris*, and other problematic, taxa (eg, Centrocoris), were only included in the *Heteroptera from the Iberian Peninsula* group's dataset when the species determination was backed-up by a physical specimen identified in the lab.

3.5 Developing faunistic catalogs and datasets

The task of developing faunistic catalogs begins by recognizing that taxa possess names that have changed, and will continue to change, over time. Therefore, it becomes imperative to establish a base taxonomic nomenclature for the taxa under consideration. An effective method is to use the nomenclature proposed in a well establish catalog of the region under consideration (e.g., Catalogue of the Heteroptera of the Palaearctic Region, Zoological Catalogue of Australia), a practice that we followed in this work. This base nomenclature was then updated by researching more recent entomological published papers, from which the changes in the names of taxa, new regional occurrences of species, invasion of alien species, and the description of new ones, was documented. Data used to establish this updated nomenclature also served to elaborate a 'thesaurus' of species synonyms, which was instrumental in the appropriate placement of taxa found under different names in the older literature.

After establishing the updated taxonomic nomenclature and thesaurus of species synonyms, we proceeded to systematically search the entomological literature for records of heteropteran bugs. We attempted to cover any published work done

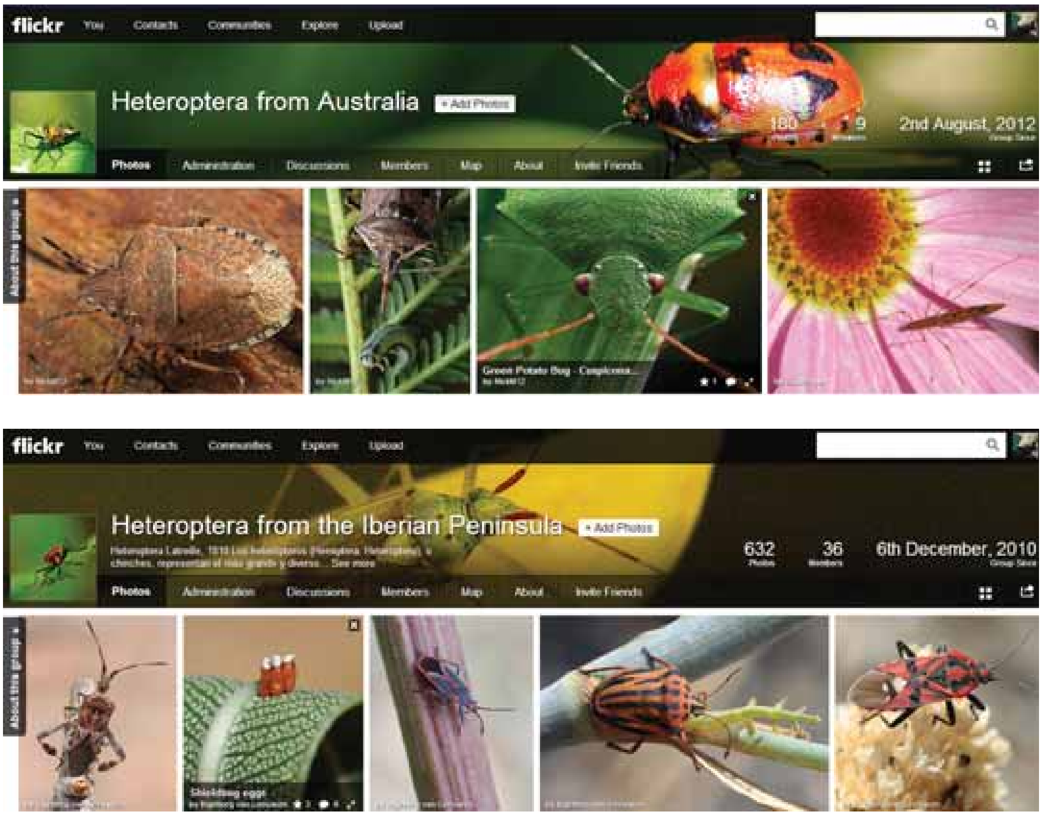

Figure 3.3 Screenshots from the Flickr groups *Heteroptera from Australia* (Top) and *Heteroptera from the Iberian Peninsula* (Bottom). Source: http://www.flickr.com/groups/australianbugs/ and http://www.flickr.com/groups/iberianbugs/

by entomologist working in Victoria, Australia or the Iberian Peninsula. Besides documenting the species name, authorship, year of description, and taxa affiliation, we also assigned, whenever possible, each record to at least one of a series of hierarchical spatial units. These units coincide generally with administrative levels, but may also be related to conservation spaces, such as Natural/National Parks or geographic units that span more than one administrative level. For example, a record from the Iberian locality of *Can Busquets* was assigned to *La Floresta* (town), *San Cugat del Vallès* (municipality), *Vallès Occidental* (shire), *Barcelona* (province), *Catalonia* (autonomous state) and *Spain* (country), but also to *Collserola Natural Park* (a large conservation space spanning several municipalities).

To complement the bibliographic records we documented the photographic records of at least two biodiversity web resources: Biodiversidad Virtual and the Flickr groups *Heteroptera from Australia* and *Heteroptera from the Iberian Peninsula*. On specific occasions we also documented the records found in The Encyclopedia of Life. A full description of these web resources is given in *Photographic biodiversity web resources* above. In order to guarantee their correct taxa assignation, an attempt was made to thoroughly curate each photographic record included in the web resources' datasets.

Finally, additional records were provided by new unpublished material stemming from a series of field insect surveys. The larger proportion of these records came from field surveys either conducted directly by the author or in collaboration with other

Figure 3.4 The pentatomid *Carpocoris fucispinus* (Boheman, 1851) photographed in Alcossebre (Castellón, Valencia, Spain). Source: original.

researchers. Others were provided by surveys that were part of larger biodiversity assessments or related biodiversity research projects. In these cases, field work was conducted by the author or other researchers, and the specimens were made available to the author for sorting and identification, and, sometimes, for processing into private and/or public collections. Further records were also provided by surveys conducted entirely by other entomologists; in these cases, specimens were either provided to the author for sorting and identification or the species record data were directly provided in written or digitized format. Overall, a large amount of the species data provided by all these surveys constituted the basis for developing the state variables used in the various ecological models presented in this work (See *Case studies*). See Table M1 in the Supplementary materials for a summary of these sources.

In some cases we were interested in developing distribution maps. For this, we first generated a base map of the study area of interest (e.g., Iberian Peninsula) by downloading open-access shapefiles from the Global Administrative Areas spatial database (Global Administrative Areas 2013), for the necessary administrative levels (e.g., country, district, province and/or municipality), and merged them using a GIS (ArcGIS version 10.1). This base map was loaded into R (R Development Core Team 2012) using package `maptools` (Lewin-Koh & Bivand 2012). We then generated maps, by calling the method 'spplot' in package `sp` (Pebesma & Bivand 2005), which plotted the maps filling the polygons (i.e., administrative levels) according to the species occurrence or diversity data contained in our datasets.

PORTUGAL

Aveiro (7)
Beja (18)
Braga (3)
Bragança (5)
Castelo Branco (11)
Coimbra (10)
Évora (16)
Faro (19)
Guarda (9)
Leiria (12)
Lisboa (13)
Portalegre (15)
Porto (6)
Santarém (14)
Setúbal (17)
Viana do Castelo (2)
Vila Real (4)
Viseu (8)

Álava (28)
Albacete (56)
Alicante (59)
Almería (50)
Asturias (24)
Ávila (39)
Badajoz (42)
Baleares (67)
Barcelona (64)
Burgos (33)
Cáceres (41)
Cádiz (47)
Cantabria (25)
Castellón (57)
Ciudad Real (55)
Córdoba (45)
Cuenca (54)
Gerona (65)
Granada (49)
Guadalajara (52)
Guipúzcoa (27)
Huelva (43)
Huesca (60)
Jaén (46)
La Coruña (20)
La Rioja (29)
León (31)
Lérida (63)
Lugo (21)
Madrid (40)
Málaga (48)
Murcia (51)
Navarra (30)
Orense (23)
Palencia (32)
Pontevedra (22)
Salamanca (38)
Segovia (36)
Sevilla (44)
Soria (37)
Tarragona (66)
Teruel (62)
Toledo (53)
Valencia (58)
Valladolid (35)
Vizcaya (26)
Zamora (34)
Zaragoza (61)

SPAIN

Figure 3.5 In this work we considered the Iberian Peninsula bioregion to be divided into 67 spatial units: Andorra (1), the 18 continental districts of Portugal (2-19), and the 47 continental provinces of Spain (20-66) plus the Balearic Islands (67). Source: original

3.5.1 *Catalog of the Heteroptera from the Iberian Peninsula*

In this example we were interested in elaborating a faunistic catalog for the Iberian Peninsula heteropterofauna. We began by extracting the taxonomic nomenclature proposed in the Catalogue of the Heteroptera of the Palaearctic Region (Aukema and Rieger 1995, 1996, 1999, 2001, 2006, Aukema et al. 2013). This base nomenclature was updated by a series of more recent entomological papers, which documented changes in the names of taxa, new regional occurrences of species, invasion of alien species, and the description of new species. To find records of heteropteran bugs from the Iberian Peninsula, we searched the entomological literature, spanning the timeframe between the years 1800 and 2013. As explained in *Developing faunistics catalogs and datasets*, we are interested in assigning these records to spatial units. Here, we considered the bioregion to be divided into 67 spatial units: Andorra, the 18 continental districts of Portugal, and the 47 continental provinces of Spain plus the Balearic Islands (Figure 3.5). The British overseas territory of Gibraltar and the French area known as French Cerdagne, which account for approximately 0.1% of the Iberian Peninsula territory, were not considered part of the study area. In the few cases when a species had been recorded in Gibraltar it was assigned to the Spanish province of Cádiz. Bibliographic records were complemented with in-situ photographic records, which were curated from the photographic databases of Biodiversidad Virtual and the Flickr group *Heteroptera from the Iberian Peninsula* (see *Photographic biodiversity web resources*). New records for this catalog came from: (1) field work conducted by the author from June 2007 and March 2013, (2) unpublished records provided by a series of biodiversity assessments and projects (see Table M1 in the Supplementary materials), and (3) specimens provided by friends and collaborators (see Acknowledgments). Some of the specimens collected by the author have become part of his collection, while thousands of specimens have been placed in the public collection of the Centre for Animal Biodiversity Resources (CRBA - University of Barcelona).

3.5.2 *Heteroptera from El Maresme*

In this example, we sought to document the heteropterofauna of El Maresme shire, north-eastern Iberian Peninsula (Figure 3.6). El Maresme is located between the Mediterranean Sea and the Sant Mateu, Corredor and Montnegre Massifs, and has an approximately surface of 400 km^2. We considered the region to be divided in 30 spatial units, one for each of the shire's municipalities, and used the same taxonomic nomenclature established for the *Catalog of Heteroptera from the Iberian Peninsula* described above, as well as the same methodological procedures to locate bibliographical and photographic records. Almost all new heteropteran bugs field records were provided by an insect survey completed in 2011 from 18 March to 24 May (see *Effects of urbanization on occupancy and species richness*). Other unpublished records were also included, this were labeled as 'Other material studied'.

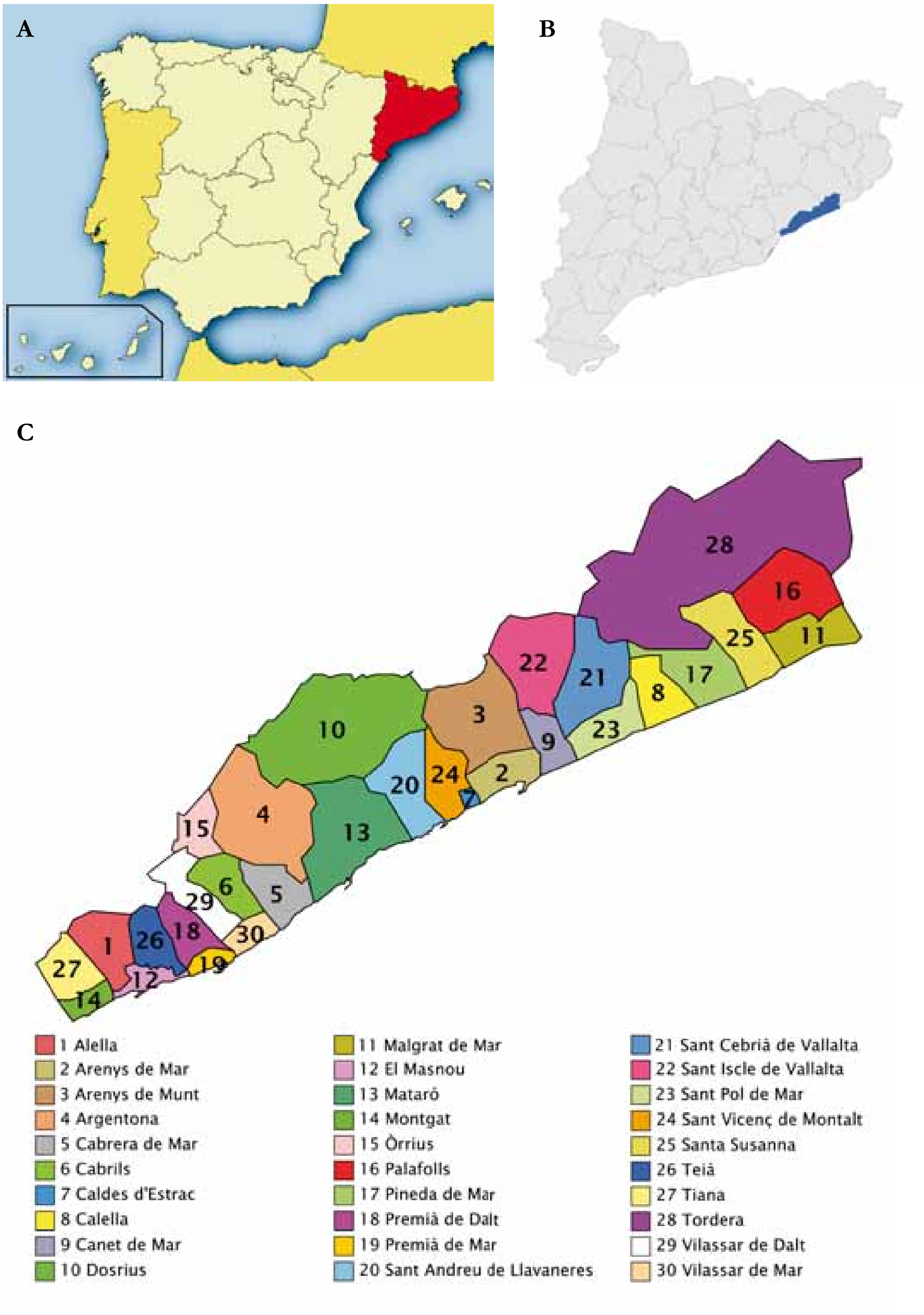

Figure 3.6 **A.** Location of Catalonia (red) within the Iberian Peninsula. **B.** Location of El Maresme shire (blue) within Catalonia. **C.** The 30 municipalities constituting El Maresme shire. Source: Wikipedia Commons.

Figure 3.7 A brachypterous firebug *Pyrrhocoris apterus* (Linnaeus, 1758) photographed in Premià de Mar (Barcelona, Catalonia, Spain). Source: original

3.5.3 *Pyrrhocoridae from the Iberian Peninsula*

This example was taken from a larger synthesis by the same name conducted by L. Mata, J.M. Grosso-Silva and M. Goula (submitted manuscript). This work undertook a general review of the state of knowledge concerning the heteropteran family Pyrrhocoridae in the context of the Iberian Peninsula bioregion. The review included aspects of pyrrhocorid taxonomical diagnosis, contemporary systematics, general biology and geographic distribution. In the present example, however, we focused only in the geographic distribution of the family, as elucidated through the methodology described in *Developing checklists and catalogs*. As in the *Catalog of Heteroptera from the Iberian Peninsula* example, the bioregion was considered to be divided in 67 spatial units: Andorra, the 18 continental districts of Portugal, and the 47 continental provinces of Spain plus the Balearic Islands (Figure 3.5). The British overseas territory of Gibraltar and the French Cerdagne were not considered part of the study area, and species recorded in Gibraltar were assigned to the Spanish province of Cádiz. To find records of the two pyrrhocorid species described in the bioregion, *Pyrrhocoris apterus* (Linnaeus, 1758) (Figure 3.7 and A3.7C) and *Scantius aegyptius* (Linnaeus, 1758) (Figure 3.8), we searched the entomological literature spanning the timeframe between the years 1877 and 2012. To complement records stemming from the literature, we curated 284 photographs from The Encyclopedia of Life, Biodiversidad Virtual and the Flickr group *Heteroptera from the Iberian Peninsula*. New specimens and observations presented in this example were collected between April 1996 and March 2013 by either J.M. Grosso-Silva, at the Centro de Investigação em Biodiversidade e Recursos Genéticos, Universidade do Porto, or the author. Distribution maps for *P. apterus* and *S. aegyptius* were elaborated following the methodology described above.

3.5.4 Heteroptera from Victoria

Our aim in this example was to build a dataset to document the heteropteran bug species occurring in Victoria, Australia (Figure 3.9). To elaborate the dataset, we first extracted the taxonomic nomenclature proposed in the Zoological Catalogue of Australia (Cassis and Gross 1995, 2002), and documented the species occurring in Victoria. This base dataset was then updated by a series of more recent entomological papers, which documented changes in the names of taxa and new occurrences of species for Victoria. We then explored The Encyclopedia of Life and the Flick group *Heteroptera from Australia* for photographic records. We finally complemented these data with new field records stemming from an insect survey completed in 2012 within south-eastern Melbourne (see *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*).

3.6 Developing new diagnostic dichotomous keys

To develop new diagnostic dichotomous keys we completed the following steps. We began by documenting all relevant keys published in the entomological literature for the taxon for which we wished to develop a new key. For taxa occurring in the Iberian Peninsula, the most important sources of robust and well tested keys are given in Table M2 (Supplementary materials). For Australian taxa, we followed the Hemiptera chapter of "The Insects of Australia" (Carver et al. 1991) and

Figure 3.8 One of the subspecies of *Scantius aegyptius* (Linnaeus, 1758). Source: Juan Manuel Sesma (Biodiversidad Virtual)

innumerable contributions found in the works of G. Cassis, G. Gross, M. Malipatil and many others.

In a second step, we wrote down all the taxa contained within the taxon we wished keyed. For this, we used the *Catalog of Heteroptera from the Iberian Peninsula* and the dataset *Heteroptera from Victoria*. As previously explained, both faunistic works were prepared with the most recent available bibliographical, photographical and field records. Thus, we are confident our keys are up-to-date with the latest taxonomical knowledge of the taxa considered.

We then proceeded to identify a set of optimal morphological characters and characters states to construct our keys, including the formal identification of exceptional character states. To choose among the vast amount of available characters for heteropteran bugs we followed two basic rules of thumb: observability and reliability (Quicke 1993). A description of the characters, and their states, used in the present work is given below. We paid special attention to characters that were equally efficient in both 'under the microscope' and 'photographic' identification. Also, we specifically acknowledged the difficulties that arise when certain species present exceptional characters states. Whenever appropriate, information regarding these exceptions was incorporated into the keys as endnotes. Other times, when the exceptions were to numerous, we developed a separate alternative key just for that character. For example, when developing our *Key to the genera of Rhyparochrominae from the Iberian Peninsula* we documented that 18 genera included one or more species that present brachypterous forms or were exclusively brachypterous. Hence, for these brachypterous taxa a separated key was developed.

Next, we wrote down the keys following an intermediate technique in which we used one or more characters per dichotomous couplet. In other words, we developed our keys using a combination of the mono- and polythetic methods (Pankhurst 1978). Overall, 'branching couplets' leading to other couplets were more general in nature, whereas 'terminal leads' leading to species or taxon identification were much more specific and rich in information. For example, in *Key to the families of Heteroptera from the Iberian Peninsula* the first lead reads:

1. Antennae longer than head **3**

whereas the lead that allows the identification of the Microphysidae reads:

10. Body length between 1.1 and 2.4 mm. ♀: Brachypters. Head without ocelli. Abdomen globular. Myrmecomorphs. ♂: Macropters. Head presenting ocelli. Hemelytra presenting cuneus .... **Microphysidae** Dohrn, 1859

Moreover, if a taxon identified in a terminal lead was represented in the region of interest by a single species, then nomenclature information about it was also provided. Here is an example extracted from the same key mentioned above:

— Rostrum reaching at least metacoxa. Coria of the macropterous forms (♀) without indentation. Living on moss .... **Ceratocombidae** Fieber, 1860

One species: *Ceratocombus coleoptratus* (Zetterstedt, 1819)

We were also interested in including in our key development methodology the use of logical connectives (Enderton 2001). This allowed us to develop logical couplets by compounding characters or character states. Various words or word pairs expressing logical connectedness, including and for conjunction and or for disjunction, were

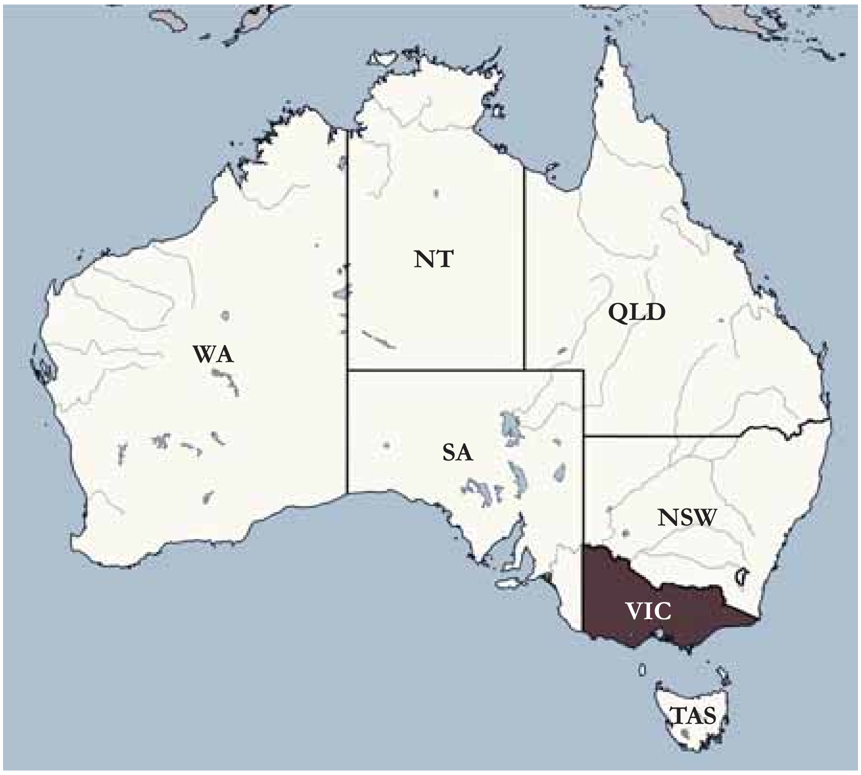

Figure 3.9 Australia. The shaped area respresents the State of Victoria (VIC). Other states: Western Australia (**WA**), Northern Terriroty (**NT**), South Australia (**SA**), Queensland (**QLD**), New South Wales (**NSW**) and Tasmania (**TAS**). Source: Wikipedia Commons.

used when appropriate, and, as just shown, were underlined in the text for clarity. For example, the first lead of the third couplet in *Key to the tribes of Miridae form the Iberian Peninsula* states:

> 3. Antenomers III are the longest antennal segments and tarsomers III are the longest tarsal segments . . . . **Bryocorini**

which should be interpreted, following logical reasoning, that the specimen under identification belongs to the Bryocorini if and only if both the sentences before and after the and are true simultaneously. In opposition, the second lead:

> — Antenomers III are not the longest antennal segments and/or tarsomers III are not the longest tarsal segments **4**

implies that as long as one of the two sentences is correct the specimen is not a Bryocorini, and identification must move forward to couplet number 4.

We note in passing, that polythetic couplets are in fact logical couplets in which the conjunction operand and has been omitted.

We followed the steps described above to obtain working drafts of our keys. These, were thoroughly tested using collection and/or new field material. When the latter was used, the correct placement of each taxa identified by the draft key was verified with well establish heteropteran bug taxonomical literature. On occasions (sometimes more frequently that I would have liked), drafted keys lead to incorrect taxonomical placements. Fortunately, these occasions led to the identification of sources of error and to essential modification of our keys.

In a final step, keys were illustrated with drawings and photographs. We went into lengths to include photographic material, either in the form of in-situ photographs of particular species or as microphotographs that highlighted a specific detail of the taxa under consideration. To this purpose, we used our own material, but also open-access photographic material available in various biodiversity web resources (see *Photographic biodiversity web resources*).

Below we provide the morphological definitions and characters used to develop our diagnostic keys. It is important to note that these characters, or character states, only apply to the mature stage of the taxa under consideration. All measurements were made in millimeters.

Body

Body organized in head, thorax and abdomen. Body length measured from apex of clypeus to apex of abdomen.

Head

Head length measured from apex of clypeus to margin dividing head and pronotum. Head width, also denominated diatone, measured from external margin of left eye to external margin of right eye. Antennae 4- or 5-segmented, segments denominated antenomers. Rostrum 3- or 4-segmented. Compound eyes always present. Ocelli present or absent. A transversal furrow might be present between compound eyes and ocelli. The clypeus is an anterior sclerite of the head, which delimits the head's dorsum with the rostrum.

Figure 3.10 In the scutellerid *Odontotarsus purpureolineatus* (Rossi, 1790), the hemelytra are completly covered by the scutellum. Source: original (Gerona, Catalonia, Spain).

Thorax

Thorax organized in pro-, meso- and metathorax. Dorsum of prothorax denominated pronotum. Longitudinal hulls and/or transversal furrows might be present on pronotum. Dorsum of mesothorax denominated scutellum. Scutellum may reach the apex of abdomen, covering partially or completely the hemelytra (Figure 3.10). Diagonal hulls might be present on scutellum. Lateral sclerites of metathorax denominated metapleura. Metapleura generally marked by the openings of the metathoracic scent-glands. Thoracic appendages in the form of legs and wings.

Legs

Legs 5-segmented. Leg segments denominated from base to apex: coxa, trochanter, femur, tibia and tarsus. According to their position in the anterior, middle or posterior pair of legs these segment are given a pro-, meso- or meta-prefix. Tarsi 2- or 3-segmented. Tarsal segments denominated tarsomers. Pretarsi presenting claws. Ungitractor plates of claws may present parempodia.

Wings

Forewings, denominated hemelytra, highly polymorphic. Macropterous hemelytra reach approximately the apex of abdomen, and present a clear division between its anterior regions, which are generally well sclerotized, and its posterior regions, which are membranous (Figure 3.11). The anterior region of macropterous hemelytra is divided in clavus and coria. The coria is further divided into endo- and exocoria. Towards its apex, the corial lateral margin might present a transversal fracture (if present then the region between the fracture and the apex is denominated cuneus). The posterior membrane might present longitudinal and/or diagonal veins, which can join together to form cells. Brachypterous hemelytra reach at most the abdominal sternite VII, their membranes are distinctly reduced and the suture limiting the clavus and coria is less marked or absent (Figure 3.7 and A3.7C). Micropterous hemelytra reach approximately the base of abdomen, they present no membrane and there is no distinction between clavus and coria (Figure A2.28 and A3.3C). Coleopteorus hemelytra resemble coleopteran forewings (i.e., elytra), they reach the apex of abdomen and present no membranous posterior region. Hindwings are always membranous. Apterous forms present neither fore- nor hindwings (Figure 3.12).

Abdomen

Abdomen showing 7-8 visible segments. Dorsal region of each segment denominated terga, ventral region denominated sterna. Terga and sterna may be composed of one or more tergites and strenites, respectively. Last abdominal tergites and sternites modified into genitalia. In a few aquatic species, the abdominal apical appendages are modified into a respiratory siphon.

3.6.1 *Key to the families of Heteroptera from Victoria*

In this example, we present a key to the families of heteropteran bugs specifically adapted to an administrative area within the Australasia ecozone. This key follows the taxonomical revision of the Lygaeoidea done by Henry (1997). Therefore, the following taxa area included with family status: Artheneidae, Blissidae, Cryptorhamphidae, Cymidae, Geocoridae, Henicocoridae, Heterogastridae, Ninidae, Oxycarenidae, Pachygronthidae and Rhyparochromidae.

3.6.2 *Key to the families of Heteroptera from the Iberian Peninsula*

In this example, we present an updated English version of our own "Clave de Familias de Heterópteros de la Peninsula Ibérica" (Mata and Goula 2011), which in turn was inspired by an unpublished key that J. Ribes made available to the authors and by the family key developed by Vázquez (2004). Here, we provide for the first time in the context of the Iberian Peninsula a key that follows the taxonomical revision of the Lygaeoidea done by Henry (1997). As a consequence of this revision, seven Lygaeid subfamilies gained family status, including Artheneidae, Blissidae, Cymidae (Figure A3.7A), Geocoridae (Figure 4.12 and A3.6A), Heterogastridae (Figure A3.6B), Oxycarenidae (Figure 4.9 and A3.7B) and Rhyparochromidae (Figure A3.5A and all A5 Figures). We also incorporated the new family status of the Aepophilidae, as documented in Aukema and Rieger (1995).

Regarding exceptional character states, it should be noted that it was not possible to define a character or character state that applied to both macro- and brachypterous forms to separate the Coreoidea families from many of the Lygaeoidea families. Hence, as can be seen in couplet 21 below:

21. Membranes presenting at least 6 longitudinal veins **22**

— Membranes presenting at most 5 longitudinal veins **25**

we decided on a character state that only works for the macropterous forms, but included an appendix with an alternative key that allows the direct identification of 30 brachypterous taxa that were left out in this couplet.

3.6.3 *Key to the tribes of Miridae from the Iberian Peninsula*

The key presented in this example was based on the following four works: (1) 'Familia Miridae: Cláve de subfamilias' [Family Miridae: Key to subfamilies] found in Goula (1986), (2) 'Familie Miridae: Bestimungschuseel fur die Unterfamilien' [Family Miridae: Key to subfamilies] in Wagner (1974), (3) 'Key to Subfamilies of Miridae' in Schuh and Slater (1995), and (4)'Key to the Subfamilies of Miridae Known in Australia' in Carver et al. (1991). Here, as in the latter work, we have preferred to work with mirid tribes, a suprageneric level that, compared to the mirid subfamilies,

Figure 3.11 The lygaeid *Spilostethus saxatilis* (Scopoli, 1763) is a characteristically macropterous species presenting hemelytra that reach approximately the apex of abdomen. Note the sharp distinction between the well-sclerotized anterior regions of the hemelytra and the posterior overlapped membranes. Source: original (Valle del Lago, Somiedo Natural Park, Asturias, Spain).

has, historically, experienced fewer changes to their diagnostic characteristics. 'Key' to the elaboration of this key was the recent revision of the family Miridae done by Cassis and Schuh (2012).

3.6.4 *Key to the genera of Rhyparochrominae from the Iberian Peninsula*

In developing this key we were interested in solving a practical impediment that has historically accompanied the identification of Rhyparochromid species. The solution we propose here is intended for species occurring in the Iberian Peninsula, but we believe the methodology could be equally applied to other bioregions. Traditionally, identification of Rhyparochromid species begins by keying the specimen of interest in a 'Key to the tribes of Rhyparochromi…', see for example the keys in Schuh and Slater (1995), Péricart (1998) or Carver et al. (1991). The problem arises because these keys strongly rely on the use of characters and/or character states that suffer from a low degree of observability (e.g., trichobothria and spiracles). This 'natural keys' have a high degree of phylogentic coherence (Pankhurst, 1978), but may be impractical for identification purposes. Here, we followed a strictly taxonomical approach, and propose a key that allows identification directly to the genus level.

Rhyparochromids in the Iberian Peninsula presented us with yet a second challenge: 19 out of 54 genera present one (or more) species with brachypterous forms or species that are exclusively brachypterous. We circumvented this issue by developing different keys for the macro- and brachypterous forms. This allowed us to avoid the use of character or character states relating to the tergites in the former and to the hemelytra in the latter.

Finally, we note the interesting inclusion in this key of the genus Tempyra (Rhyparochrominae: Udeocorini). This stems from the recent records of *Tempyra biguttula* Stal, 1874, in the Spanish provinces of Almería, Cádiz, Córdoba and Murcia (Baena and Torres 2012, Biodiversidad Virtual 2013), which represents the first records of the species, genus and tribe for both the Iberian Peninsula and Palearctic region (Aukema et al. 2013).

3.6.5 *Key to the species of Deraeocoris from the Iberian Peninsula*

As an example of a dichotomous key that reaches down to the species level, we present here in the context of the Iberian Peninsula a key to the mirid deraeocorid genus *Deraecoris* (Kirschbaum, 1856). The key is based both on keys found in Wagner (1974) and Goula (1986). We note the interesting inclusion of *Deraecoris flavilinea* (Costa, 1862) (Figure A6.3), a fairly new addition to the Iberian Peninsula heteropterofauna (Gessé 2011, Goula and Mata 2011b, Vivas 2012).

Figure 3.12 An undescribed adult apterous heteropteran bug species photographed in a cleared tropical savanna (Katherine, Northern Territory, Australia). Source: original.

3.7 Multi-species site occupancy model (msSOM)

What follows is a multi-species extension of the hierarchical linear model that MacKenzie et al. (2002, 2005) described to estimate site occupancy rates of species that are imperfectly detected and/or of rare species with low occurrence probabilities. In a msSOM collective community data inform on the occurrence probability for all observed species, including rare species, resulting in an improved analysis of the community and increased precision in species-specific estimates of occurrence (Dorazio and Royle 2005, Dorazio et al. 2006, Kéry and Royle 2008, Zipkin et al. 2009, 2010).

Let site-specific occupancy $z(i,j)$ be defined as a binary variable for species $i = 1,2,\ldots N$ at site $j = 1,2,\ldots J$, where $z(i,j) = 1$ if species i is present in site j and $z(i,j) = 0$ otherwise. The model for occurrence is specified as:

$$z(i,j) \sim \text{Bernoulli}\ (\Psi_{i,j})$$

where $\Psi_{i,j}$ is the probability that species i occurs at site j.

The imperfect detection of species cofounds the estimation of $\Psi_{i,j}$, thus a formal distinction must be made between absence and non-detection. Sampling sites with $k > 1$ spatial or temporal replicates allows for this distinction by specifying a detection model:

$$x(i,j,k) \sim \text{Bernoulli}\ (\Phi_{i,j,k} \,.\, z(i,j))$$

where $\Phi_{i,j,k}$ is the detection probability of species i at site j at/during replicate k. Since $z(i,j) = 0$ when a species is not present, the model will only estimate non zero values of $\Phi_{i,j,k}$ when species i is in fact present at site j.

In the context of msSOM, the most basic model has the occurrence probability $\Psi_{i,j}$ be determined by unspecified species and site-specific effects, which are parameterized into the model on the logit-probability scale as follows:

$$\text{logit}\ (\Psi_{i,j}) = u_i + \alpha_j$$

where u_i are the species-level effects and α_i the site-specific effects on occurrence.

Likewise, the detection probability $\Phi_{i,j,k}$ is determined by unspecified species and site-specific effects, which are also parameterized on the logit scale:

$$\text{logit}\ (\Phi_{i,j,k}) = v_i + \beta_j$$

where v_i are the species-level effects and β_i the site-specific effects on detection.

The magnitude of $\Psi_{i,j}$ and $\Phi_{i,j,k}$ may be thought to be influenced by environmental and/or survey characteristics. If these covariates are available and were properly measured, then a linear combination of parameters and species and/or site-specific covariates may be substituted for u_i, v_i, α_j and β_j, accordingly.

3.8 Bayesian inference

In order to execute our Bayesian analysis two key steps were followed. We first assigned to each model parameter a prior distribution. In most cases we lacked external knowledge of the system under study. Therefore, we wanted our inference to be based on the observed data alone. This was achieved by using non-informative priors. Normal, Uniform, Bernoulli and Gamma probability distributions were used as appropriate. For example, to specify our lack in knowledge regarding the hyper-parameters (mean and precision) of a linear predictor's normally-distributed coefficient

$a \sim N(mu, tau)$

we used

$mu \sim N(0, 0.001)$ and

$tau \sim \text{Gamma}(0.1,0.1)$.

Secondly, we used a simulation-based technique denominated Markov chain Monte Carlo (MCMC) to draw samples from the posterior distribution of our parameters (McCarthy 2007, Kéry 2010). Here, we used the MCMC algorithms as implemented in the software OpenBUGS (see *Software and implementation* below). Values to be passed down to the MCMC algorithm included initial values for the parameters, number of chains to be run, number of iterations (i.e., draws from the posterior distribution), number of transition phase burn-in iterations and thinning rate.

In most cases we defined a function to generate random starting values. However, in other occasions, we explicitly supplied these initial values for each chain requested. Most models were ran using two or three chains, and an optimal number of iterations that guaranteed convergence was set by trial and error. A variable number of 'transition phase' draws from each chain may not be representative of the stationary distribution, thus they were discarded (i.e., burnt-in) as appropriate. Finally, depending on the complexity of the model, we sat a thinning rate, which limited the number of draws that were saved from each chain.

Before making inferences from the posterior distribution, we ensured that an equilibrium distribution had been reached by the MCMC. This convergence check was based on the Gelman-Rubin statistic, as implemented in the OpenBugs software (see *Software and implementation* below), where it is denominated 'Rhat'. Values below 1.1 indicate acceptable convergence (Gelman and Hill 2007).

3.9 Software and implementation

All analyses were implemented using the open-access software environments R and OpenBUGS. R is a system for statistical computation and graphics (R Development Core Team 2012). Its development has been heavily influenced by the computing languages S and Scheme (Hornik 2012). R contains functionality for a large variety of statistical and graphical procedures, which can be flexibly expanded by a series of additional modules (i.e., add-on packages or libraries). OpenBUGS implements the BUGS language (Bayesian analysis Using Gibbs Sampling) to specify complex statistical models using Markov chain Monte Carlo (MCMC) techniques under the Bayesian mode of inference (Gilks et al. 1994, Spiegelhalter et al. 2012). OpenBUGS can be ran natively or accessed remotely. Here, we specified and ran all models using the latter method, using the R package `R2OpenBUGS` (Sturtz et al. 2005).

3.10 Case studies

3.10.1 *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*

Our species data were provided by an insect survey completed from 14 January to 12 March 2012 at 104 plots within 13 golf courses in the costal-plain soil-bioregion of south-eastern Melbourne, Victoria, Australia (Figure 3.13). With an area of 7,694 km^2, Melbourne is the second largest city in Australia, and has a population of approximately 4.1 million people (Australian Bureau of Statistics 2012). Centered at the estuary of the Yarra River, Melbourne is located on a large natural bay known as Port Phillip. Insects were collected at eight randomly selected independent sampling plots within each one of the 13 golf course sites. Plots, which had a surface area of 600 m^2 (20 x 30 m), were stratified by vegetation into two groups; 1) woodland, consisting of managed 'canopied-rough' (Figure 3.14), or 2) grassland, consisting of less intensively managed herbaceous 'old-rough' (Figure 3.15). At each plot,

Figure 3.13 The Melbourne Metropolitan Area (Victoria, Australia). Red dots indicate the 13 golf courses which were part of the *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* case study. Source: Google Earth (with modifications).

Figure 3.14 An example of a woodland patch within a golf course in south-east Melbourne (Victoria, Australia). Source: original.

200 sweeps of an entomological net were used to collect specimens of all insect species available for sampling. Since we selected a balanced number of plots of each vegetation group, four spatial replicates of each group per golf course were available for analyses. Heteropteran bugs were sorted out of each replicate sample and, whenever possible, identified to species.

A vegetation survey of each plot was also conducted parallel to the insect survey. Through this survey it was possible to characterize each plot's vegetation density, which we use here as a covariate influencing heteropteran bug occurrence probabilities. Vegetation density, measured as the number of times vegetation was intercepted at all heights divided by the total number of points taken, varied from 27.4 to 50.0 (mean=34.4) , ranging from 0.14 to 0.84 (mean=0.40) in woodland and from 0.14 to 0.57 (mean=0.28) in grassland.

We used an unconditional multi-species site occupancy model to estimate the species richness of the woodland heteropteran bug community, plus that of its herbivore and predatory guilds. This same model estimated woodland community-level occurrence and detection probabilities. We then ran a second model using a dataset that combined the woodland plus grassland occupancy data. Finally, we modified this latter model to incorporate the effect of vegetation density on the species-specific probability of occurrence. The model for occurrence was specified as:

$z(i,j) \sim \text{Bernoulli}\ (\Psi_{i,j})$

where $\Psi_{i,j}$ was the probability that species *i* occurred at golf course *j*. The observation model, for which we have recorded data $x(i,j,k)$ for species *i* at golf course *j* at the k^{th} plot, was specified as:

$$x(i,j,k) \sim \text{Bernoulli}\ (\Phi_{i,j,k} \cdot z(i,j))$$

where $\Phi_{i,j,k}$ was the detection probability of species *i* at golf course *j* at plot *k*. This satisfied the condition that the detection probability of a species will be zero when it is not present.

We incorporated the effect of vegetation density (VD) over the species probability of occurrence (Ψ) on the logit-probability scale as follows:

$$\text{logit}\ (\Psi_{i,j}) = u_i + \alpha 1_i(VD_j)$$

where u_i was the species-level effect and, $\alpha 1_i$ the site-specific effect on occurrence. Vegetation density was standardized so that its mean was zero and its standard deviation one. By doing this, the logit-inverse of u_i becomes the occurrence probability of the average golf course. Since the survey was completed in less than 2 months, we confidently assumed that the heteropteran bug species pool remained constant, thus satisfying an important assumption of the model. We also assumed that the detection probability of species *i* did not vary based on any measured covariate, thus it is determined by an unspecified species-level effect v_i as:

$$\text{logit}\ (\Phi_{i,j,k}) = v_i$$

We considered all occurrence and detection parameters as random effects governed by hyperparameters, and estimated the model parameters and community

Figure 3.15 An oldrough within a golf course in south-east Melbourne (Victoria, Australia). Source: original.

summaries under a Bayesian mode of inference. Models were specified and ran in OpenBUGS, as accessed through the R add-on package `R2OpenBUGS`. We used 2 chains of 100,000 iterations and discarded 10,000 as burn-in. Values of the Gelman-Rubin statistic for all parameters (R-hat < 1.01) indicated acceptable convergence. Hyperparameters were given uninformative priors, thus species-specific effects were given uniform (from 0 to 1) priors, while the mean and the standard deviation of the site-specific effects were given normal (mean zero and variance 1000) and gamma (r and nu equal to 0.1) priors, respectively. See Model 1 and Model 2 in Models (Supplementary materials) for the R and OpenBUGS model codes.

3.10.2 *The effect of landscape functional heterogeneity on vineyard biodiversity*

Species data were provided by an insect survey conducted from late April to late June 2011 at the localities of Castellet i La Gornal (CG) and Avinyó Nou (AN) within the Alt Penedès shire, north-eastern Iberian Peninsula (Figure 3.16). Insects were collected at 10 randomly selected independent sampling sites (i.e., vineyards) within each locality. Vineyard sites had a surface area of at least 1ha. A sampling system was set at each vineyard consisting of a flight intercept (FI) and a pitfall (PF) trap (Figure 3.2). These where placed towards the vineyard's center, and were separated from each other by at most 2 m. All sites were sampled on a weekly basis for 10 consecutive weeks. Thus, 20 sampling replicates for each vineyard were available for analyses. Heteropteran bug specimens were sorted out of each replicate sample, identified to species and assigned to a functional guild either as herbivores or predators.

Circular habitat maps (500 m radius buffers around the centre point of each vineyard site) provided by the BioDiVine project were used to develop a measure of functional landscape heterogeneity. The following 'natural habitat' types were considered as beneficial for heteropteran bugs: Oak forest, shrubland, meadows, oldfields and gardens. Their area data were summarized and used to calculate the proportion of natural habitat (PNH) of each site. This landscape attribute was used in our models as a covariate influencing species occurrence. Within the CG (Castellet

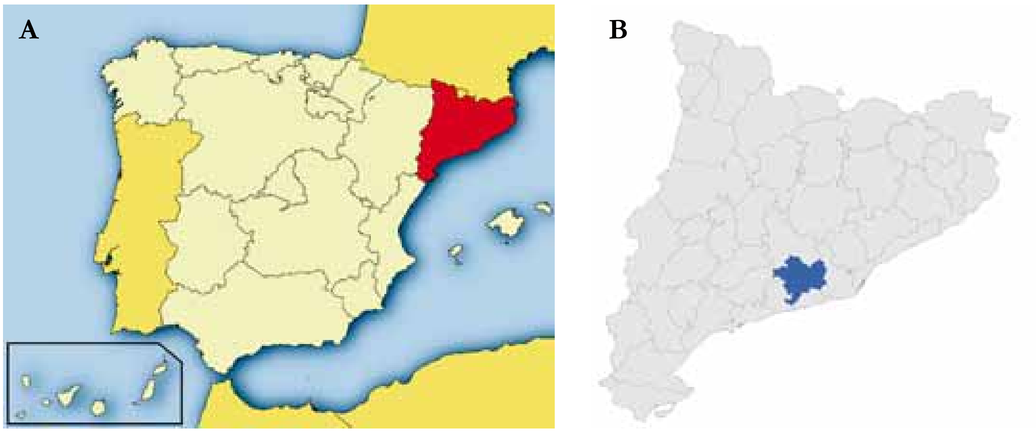

Figure 3.16 A. Location of Catalonia (red) within the Iberian Peninsula. **B.** Location of the Alt Penedès shire within Catalonia. Source: Wikipedia Commons.

i La Gornal) and AN (Avinyó Nou) study sub-regions, PNH ranged from 0.02 to 0.91 (mean=0.42).

We used a first set of multi-species site occupancy models (msSOMs) to estimate the effect of study sub-region (CG and AN) on herbivore and predatory guild species richness. This same set of models were also used to quantify the effect of sampling trap type (FI and PF) on community and species-specific detection probabilities. Taking into account only the whole study area FI data, we then ran a second set of models, in which we quantified the effect of the PNH covariate on the herbivorous and predatory heteropteran bug species-specific occurrence probabilities. In both cases, the model for occurrence was specified as:

$z(i,j) \sim \text{Bernoulli}\ (\Psi_{i,j})$

where $\Psi_{i,j}$ was the probability that species i occurred at vineyard j. The observation model, for which we have recorded data $x(i,j,k)$ for species i at vineyard j during the k^{th} replicate, was specified as:

$x(i,j,k) \sim \text{Bernoulli}\ (\Phi_{i,j,k} \cdot z(i,j))$

where $\Phi_{i,j,k}$ was the detection probability of species i at vineyard j during replicate k. This satisfied the condition that the detection probability of a species will be zero when it is not present.

Effects on the species probability of occurrence (Ψ) were specified on the logit-probability scale. In the first set of models, depending on whether site j belonged in the CG or AN sub-region, the linear predictors were defined as:

$\text{logit}\ (\Psi_{i,j}) = CG_i$

$\text{logit}\ (\Psi_{i,j}) = AN_i$

where CG_i and AN_i were the CG and AN species-level effects on occurrence, respectively. In the second set of models, we incorporated the effect of the PNH covariate on the species probability of occurrence (Ψ) as follows:

$\text{logit}\ (\Psi_{i,j}) = u_i + \alpha 1_i (PNH_j)$

where u_i was the species-level effect and, $\alpha 1_i$ the site-specific effect on occurrence. The PNH covariate was standardized so that its mean was zero and its standard deviation one. By doing this, the logit-inverse of u_i becomes the occurrence probability of the average vineyard. Since the survey was completed in less than 2 months, we confidently assumed that the Heteroptera species pool remained constant, thus satisfying an important assumption of the model.

Effects on the species detection probabilities (Φ) were also specified on the logit-probability scale. In the first set of models, depending on whether site j belonged in CG or AN and whether replicate k was sampled through a FI or PF trap, the linear predictors were defined as:

$\text{logit}\ (\Psi_{i,j}) = FICG_i$

$\text{logit}\ (\Psi_{i,j}) = FIAN_i$

$\text{logit}\ (\Psi_{i,j}) = PFCG_i$

$$\mathrm{logit}\,(\Psi_{i,j}) = \mathrm{PFAN}_i$$

where FICG_i was the CG species-effect on FI detection and FIAN_i the AN effect. And PFCG_i and PFAN_i were the species-effects on PF detection, respectively. In the second sets of models, we assumed that the detection probability of species *i* did not vary based on any measured covariate, thus it was determined by an unspecified species-level effect v_i as:

$$\mathrm{logit}\,(\Phi_{i,j,k}) = v_i$$

We considered all occurrence and detection parameters as random effects governed by hyperparameters, and estimated the model parameters and community summaries under a Bayesian mode of inference. Models were specified and ran in OpenBUGS, as accessed through the R add-on package `R2OpenBUGS`. We ran two chains of 25,000 iterations, discarded the first 2,500 and thinned by two. Values of the Gelman-Rubin statistic for all parameters (R-hat < 1.01) indicated acceptable convergence. Hyperparameters were given uninformative priors, thus species-specific effects were given uniform (from 0 to 1) priors, while the mean and the standard deviation of the site-specific effects were given normal (mean zero and variance 1000) and gamma (r and nu equal to 0.1) priors, respectively. See Model 3 and Model 4 in Models (Supplementary materials) for the R and OpenBUGS codes.

3.10.3 Effects of urbanization on occupancy and species richness

Species data were provided by a terrestrial insect survey collected from 18 March to 24 May 2011 at the largest city of each one of the 30 municipalities that constitute the shire of El Maresme in the north-east Iberian Peninsula (Figure 3.6). The shire is located between the Mediterranean Sea and the Sant Mateu, Corredor and Montnegre Massifs, and has an approximately surface of 400 km^2. It has a littoral Mediterranean climate: annual average precipitation oscillates between 550 mm at sea level and 800 mm in the mountains, and annual average temperatures vary from 8º in the winter to 23º in the summer (Servei Meteorològic de Catalunya 2010).

Insects living on herbaceous ruderal vegetation (Figure 3.17) were collected at two randomly selected independent sampling plots within each city, however due to habitat unavailability in seven cases it was only possible to collect from one plot. Sampling plots were located at least: (1) 150 m away from the city center, (2) 150 m away from the boundary with another city, and (3) 500 m apart from each other. At each plot, the author or a trained field researcher used 100 sweeps of an entomological net to collect at least one specimen of each insect species available for sampling. All plots were visited twice during the survey's duration, thus two to four sampling replicates per each city were available for analyses. Heteropteran bugs were sorted out of each replicate sample and identified to species.

For each city, we transformed aerial photographs (Institut Cartogràfic de Catalunya 2011) in a GIS environment (ArcGIS version 10.1) into circular land-use maps (r=750 m). This is the maximum radius that a city in El Maresme can have before it overlaps into the boundaries of another. We quantified each city's degree of urbanization through the Weeks' index of urbanization (Weeks et al. 2005). The index, which is set to a 0 to 100 scale, combines land-use with census data to generate

Figure 3.17 Examples of ruderal herbaceous vegetation plots in El Maresme shire (Barcelona, Catalonia, Spain) that were part of the *Effects of urbanization on occupancy and species richness* case study. Top: Dosrius. Bottom: Palafolls. Source: original (top photo with permission of Helena Caselles and Josep Solà).

an urbanization gradient where higher values are associated with higher degrees of urbanization. In our study the census data were provided by the Institut d'Estadística de Catalunya (2010). With a mean value of 51, the index in our study area ranged from 27 in the interior rural municipality of Òrrius (Figure 3.18, top) to 79 in the shire's capital city of Mataró (Figure 3.18, bottom).

To estimate the species richness of the whole heteropteran bug community, as well as that of its herbivorous and predatory guilds, and to quantify the effects of urbanization on the species-specific occurrence and detection probabilities, we used a multi-species site occupancy model (msSOM). The model for occurrence was specified as:

$z(i,j) \sim \text{Bernoulli}(\Psi_{i,j})$

where $\Psi_{i,j}$ is the probability that species i occurs at city j. The observation model, for which we have recorded data $x(i,j,k)$ for species i at city j at the k^{th} spatial or temporal replicate, is specified as:

$x(i,j,k) \sim \text{Bernoulli}(\Phi_{i,j,k} \cdot z(i,j))$

where $\Phi_{i,j,k}$ is the detection probability of species i at city j at/during site/period k. This satisfies the condition that the detection probability of a species will be zero when it is not present.

We incorporated the effect of urbanization (URB) over the species probability of occurrence (Ψ) on the logit-probability scale as follows:

$\text{logit}(\Psi_{i,j}) = u_i + \alpha 1_i(\text{URB}_j)$

where u_i is the species-level effect and, $\alpha 1_i$ is the site-specific effect on occurrence. URB was standardized so that its mean was zero and its standard deviation one. Since the survey was completed in about 2 months, we confidently assumed that the Heteroptera species pool remained constant, thus satisfying an important assumption of the model. We also assumed that the detection probability of species i did not vary based on any measured covariate, thus it is determined by an unspecified species-level effect v_i as:

$\text{logit}(\Phi_{i,j,k}) = v_i$

We considered all occurrence and detection parameters as random effects governed by hyperparameters, and estimated the model parameters and community summaries under a Bayesian mode of inference. Models were specified and ran in OpenBUGS, as accessed through the R add-on package `R2OpenBUGS`. We used 2 chains of 30,000 iterations, discarded the first 3,000 iterations as burn-in, and thinned by 2. Values of the Gelman-Rubin statistic for all parameters (R-hat < 1.01) indicated acceptable convergence. Hyperparameters were given uninformative priors, thus species-specific effects were given uniform (from 0 to 1) priors, while the mean and the standard deviation of the site-specific effects were given normal (mean zero and variance 1000) and gamma (r and nu equal to 0.1) priors, respectively. See Model 5 in Models (Supplementary materials) for the R and OpenBUGS code.

3.10.4 *Estimation of species and family detectability along macroecological gradients*

Species data were provided by the *Catalog of Heteroptera from the Iberian Peninsula*. This catalog held distributional data for the 1,470 species and subspecies comprising the Iberian Peninsula heteropterofauna. As reported below in the results section of the present work, the core data for this catalog were extracted from 14,082 bibliographical records atomized throughout 200 years of Iberian and Palearctic entomological literature. These core data were complemented by (1) more than 23,000 photographic records provided by Biodiversidad Virtual and the Flickr group *Heteroptera from the Iberian Peninsula* (see *Photographic biodiversity web resources*), and (2) approximately 3,500 new field records contributed by the present thesis.

Figure 3.18 Aerial photographs of the Òrrius (top) and Mataró (bottom) municipalities (Barcelona, Catalonia, Spain). Red circles indicate the study site area that was considered in the *Effects of urbanization on occupancy and species richness* case study. Source: Insitut Cartogràfic de Catalunya (2013).

As the westernmost peninsula in southern Europe, the Iberian Peninsula is bordered by the Atlantic Ocean and Mediterranean Sea, and separated from the rest of the Eurasian continent by the natural border constituted by the mountain range known as the Pyrenees. With an area of approximately 582,000 km^2, it is the second-largest peninsula in Europe, with a population of approximately 58 million people (see Table M3 in the Supplementary materials for the sources used to estimate these data). As defined previously, we considered the bioregion to be divided into 67 spatial units: Andorra, the 18 continental districts of Portugal, and the 47 continental provinces of Spain plus the Balearic Islands (Figure 3.5). Here, I treated these spatial units as inference points. Records assignable to one of these spatial units were used to construct species by point matrices for each non-monospecific family. To

Table 3.1 Area (**A**), altitudinal range (**AR**), mean annual temperature (**MAT**), mean annual precipitation (**MAP**) and population density (**PD**) of each spatial unit considered as inference points in the case study *Estimation of species and family detectability along macroecological gradients.*

Spatial unit	A	AR	MAT	MAP	PD
Álava	3037	1370	11.6	756	93
Albacete	14,924	1487	13.9	365	45
Alicante	5817	1558	18.0	331	190
Almería	8775	2606	18.8	200	48
Andorra	468	2106	8.2	867	114
Asturias	10,604	2648	13.5	987	102
Aveiro	2801	78	15.6	944	232
Ávila	8050	2296	10.7	403	23
Badajoz	21,766	1000	16.8	457	31
Baleares	4992	1445	17.4	531	154
Barcelona	7728	2531	15.8	613	609
Beja	10,263	253	16.9	558	17
Braga	2706	550	15.0	1449	277
Bragança	6599	1162	12.7	773	25
Burgos	14,292	1971	10.4	558	36
Cáceres	19,868	2329	16.2	527	21
Cádiz	7436	1654	17.9	602	120
Cantabria	5321	2613	14.3	1212	124
Castellón	6632	1813	17.3	458	71
Castelo Branco	6627	1106	15.9	783	34
Ciudad Real	19,813	1011	15.0	409	26
Coimbra	3974	490	16.0	675	108
Córdoba	13,771	1535	17.8	568	51
Cuenca	17,140	1359	12.9	518	24
Évora	7393	291	16.5	585	24
Faro	4997	400	17.9	509	72
Gerona	5910	2913	14.6	707	74
Granada	12,647	3479	15.4	367	60
Guadalajara	12,214	1692	10.5	497	22
Guarda	5535	844	11.2	914	34
Guipúzcoa	1980	1544	13.3	1542	363
Huelva	10,128	912	18.2	504	39
Huesca	15,636	3334	13.8	516	18
Jaén	13,496	2017	17.2	525	42
La Coruña	7951	898	13.6	1431	129
La Rioja	5045	2002	13.7	410	79

Table 3.1 Continued.

Spatial unit	A	AR	MAT	MAP	PD
Leiria	3506	404	14.7	675	123
León	15,581	1938	10.9	537	36
Lisboa	2803	227	17.4	774	628
Lérida	9856	1821	14.9	363	39
Lugo	12,172	3078	11.6	1077	33
Madrid	8028	1960	14.8	437	478
Málaga	7306	2065	18.2	537	296
Murcia	11,314	2001	18.0	303	100
Navarra	10,391	2420	12.7	702	68
Orense	7273	2084	14.7	820	61
Palencia	8052	1841	10.7	379	27
Pontevedra	4495	1177	14.3	1759	203
Portalegre	6084	777	15.7	836	22
Porto	2332	155	15.2	1237	588
Salamanca	12,350	2295	11.8	379	29
Santarém	6718	525	17.0	652	67
Segovia	6921	1695	12.0	468	47
Setúbal	5214	501	16.6	735	133
Sevilla	14,036	1129	18.9	545	90
Soria	10,306	1563	10.8	512	45
Tarragona	6303	1447	17.5	525	70
Teruel	14,810	1884	11.9	379	19
Toledo	15,370	1157	15.6	358	28
Valencia	10,806	1832	18.0	472	164
Valladolid	8111	319	12.4	439	110
Viana do Castelo	2219	822	15.2	1467	113
Vila Real	4307	1225	13.6	1023	56
Viseu	5010	697	14.0	1199	80
Vizcaya	2217	1482	14.5	1173	455
Zamora	10,561	1789	12.8	379	21
Zaragoza	17,274	2248	15.2	323	49
Minimum	468	78	8.2	200	17
Mean	8777	1520	14.7	679	116
Maximum	21,766	3476	18.9	1759	628

Units: Area (km^2), ltitudinal range (m), mean annual temperature (ºC), mean annual precipitation (mm), population density (inhab/km^2). Sources: Table M3 in the Supplementary materials construct the macroecological gradients I documented the area, altitudinal range, mean annual temperature, mean annual precipitation and population density of each spatial unit. Point values for each gradient are provided in Table 3.1, while details on how the data were documented and the sources that were used are given in Table M3 (Supplementary materials). The average spatial unit had an area of 8,777 km^2, with the smallest spatial unit corresponding to Andorra (468 km^2) and the largest to Badajoz (21,766 km^2). The spatial unit with the smallest difference between max. and min. elevations was Aveiro (78 m), whereas Granada had the largest (3,479 m). The altitudinal range for the average spatial unit was 1,520 m. Mean annual temperature varied from 8.2 ºC in Andorra to 18.9 ºC in Sevilla (mean=14.7 ºC). Almería had the lowest mean annual precipitation (200 mm), while Pontevedra showed the highest (1,759 mm). Mean annual precipitation in the average spatial unit was 679 mm. Finally, with 17 and 628 hab/km^2, the spatial units with the lowest and highest population densities were Beja and Lisboa, respectively. The average spatial unit had a population density of 116 hab/km^2.

I used a modified version of a multi-species site occupancy model to estimate species, family and whole Iberian Peninsula heteropterofauna detection probabilities.

The detection model was specified as:

$\mathrm{z}(f,i,j) \sim \mathrm{Bernoulli}\ (\Phi_{f,i,j})$

where $\Phi_{f,i,j}$ is the probability that within family *f* species *i* is detected at spatial unit *j*.

The species-level random effects on the probability of detection (Φ) were incorporated on the logit-probability scale as follows:

$\mathrm{logit}\ (\Phi_{f,i,j}) = \mathrm{u}_{f,i}$

The species-level random effects $\mathrm{u}_{f,i}$ were specified as:

$\mathrm{u}_{f,i} \sim \mathrm{Normal}\ (mu_f,\ tau_f)$

where:

$mu_f \sim \mathrm{Normal}\ (mu,\ tau)$

$tau_f \sim \mathrm{Gamma}\ (0.1,\ 0.1)$

Thus, the family-level hyperparameters were also considered random-effects governed by the global hyperparameters *mu* and *tau*, where:

$psi \sim \mathrm{Uniform}\ (0,1)$

$mu = \mathrm{logit}(psi)$

$\mathrm{t}au \sim \mathrm{Gamma}\ (0.1,\ 0.1)$

I then ran five more msSOMs, which were modified to estimate the effects of the macroecological gradients (MG). These effects were also included on the logit-probability scale as follows:

$\mathrm{logit}\ (\Phi_{f,i,j}) = \mathrm{u}_{f,i} + \alpha 1_{f,i}\ (\mathrm{MG}_j)$

where $\mathrm{u}_{f,i}$ is the species-level effect and $\alpha 1_{f,i}$ is the spatial unit-specific effect on detection. The macroecological gradients were standardized so that their mean was zero and their standard deviation one. The spatial-units effects $\alpha 1_{f,i}$ were specified as:

$\alpha 1_{f,i} \sim \text{Normal}\ (mu.\alpha 1_f, tau.\alpha 1_f)$

where:

$mu.\alpha 1_f \sim \text{Normal}\ (mu.\alpha 1, tau.\alpha 1)$

$tau.\alpha 1_f \sim \text{Gamma}\ (0.1, 0.1)$

The family-level $mu.\alpha 1_f$ and $tau.\alpha 1_f$ effect hyperparameters were governed by the global effect hyperparameters $mu.\alpha 1$ and $tau.\alpha 1$, which were specified as:

$mu \sim \text{Normal}\ (0, 0.0001)$

$tau \sim \text{Gamma}\ (0.1, 0.1)$

Model parameters were estiomated under a Bayesian mode of inference. Models were specified and ran in OpenBUGS, as accessed through the R add-on package `R2OpenBUGS`. For the unconditional models, two chains of 50,000 iterations were used, the first 5,000 iterations were discarded as burn-in, and chains were thinned by a factor of two. For the conditional models two chains of 25,000 iterations were used, the first 2,500 iterations were discarded as burn-in, and the chains were thinned by a factor of two. Values of the Gelman-Rubin statistic for all parameters (R-hat < 1.01) indicated acceptable convergence. Hyperparameters, as described above, were given uniform, normal and gamma uninformative priors as appropriate. See Model 6 and Model 7 in Models (Supplementary materials) for the R and OpenBUGS code.

4 Results

4.1 Heteropteran bug specimens

A total of 16,855 heteropteran bug specimens were collected during our entomological surveys. These translated into 4,180 new field records, 85.5 and 14.5 % in the Iberian Peninsula and Victoria, respectively. Heteropteran bugs were collected at over 400 different localities. The 334 Iberian Peninsula localities span at least half of the districts and provinces that constitute this bioregion, while the 69 Victoria localities were found exclusively in the Melbourne Metropolitan Area. Specimens were collected at 35 different habitat types (Figure 4.1). When the herbaceous-like habitats (i.e., herbaceous, field margin, urban margin, open rough, ruderal, oldfield, and grassland), were pooled into one larger 'Herbaceous-like group', they accounted for approx. 60% of all the records. The large majority of specimens were collected by sweep-net (85%), followed by those collected by flight intercept (10%), pitfall (3%) and other methods (2%).

4.2 Photographic records

4.2.1 In-situ photographs

In-situ photographs taken during our entomological surveys yielded 74 new photographic records, 68 and 6 in the Iberian Peninsula and Victoria, respectively. All photographs were machine-tagged and uploaded to the photographic community website Flickr, and, most of them, have already been picked-up by the global biodiversity web resource *Encyclopedia of Life*. Photos, as appropriate, have also been placed in the Flickr groups *Heteroptera from Australia* and *Heteroptera from the Iberian Peninsula*.

4.2.2 Biodiversidad Virtual

The original dataset provided by Biodiversidad Virtual was narrowed down to 23,015 heteropteran bug photographic records. These were contributed by 713 photographers, which documented the occurrence of 603 heteropteran bug species in 2,573 different Iberian Peninsula localities. Records included in this dataset were attributable to 94% of our working spatial units for the Iberian Peninsula. Species were identified by 11 specialist belonging to the platform's 'Heteroptera experts group'. With 1,131 photographic records (4.9% of the total), *Carpocoris fuscispinus* (Boheman, 1851) (Figure 3.4) was at the top of the seven most recorded species,

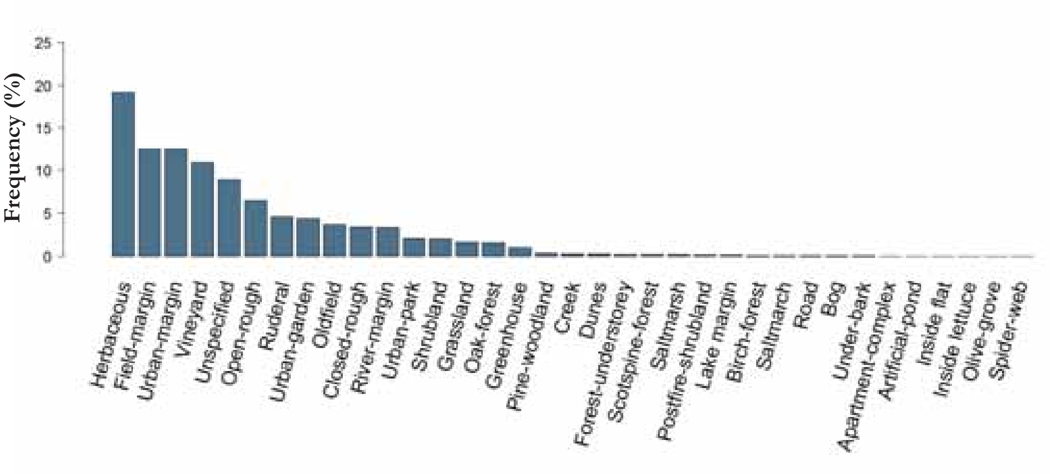

Figure 4.1 Rank by frequency plot for the 35 different habitat types in which heteropteran bugs were sampled in this work.

followed by *Nezara viridula* (Linnaeus, 1758) (733, 3.2%; Figure 4.2), *Eurydema ornata* (Linnaeus, 1758) (618, 3.0%; Figure 4.3), *Coreus marginatus marginatus* (Linnaeus, 1758) (597, 2.6%; Figure 4.4), *Dolycoris baccarum* (Linnaeus, 1758) (576, 2.5%; Figure 4.5), *Spilostethus pandurus* (Scopoli, 1763) (551, 2.4%; Figure 4.6) and *P. apterus* (530, 2.3%; Figure 3.7 and A3.7C).

I note that within the abovementioned dataset is the first record for the Iberian Peninsula of the pentatomid *Mecidea lindbergi* Wagner, 1954, which also represents the first record of the tribe Mecideini in the bioregion. This new species for the Iberian heteropterofauna was photographed in March 2011 in the Spanish province of Sevilla by M. Ramirez, and later identified by E. Ribes. To the best of my knowledge, *M. lindbergi* is known in the Iberian Peninsula bioregion exclusively thanks to this photographic record.

4.2.3 *Flickr group: Heteroptera from Australia*

A total of 24 photographs were curated from the group's pool. These were contributed by 4 photographers, which documented the occurrence of 24 heteropteran bug species in 15 different localities within Victoria, Australia. With only one exception, species were identified by the same member contributing the photograph to the group. The photographic record of the pyrrhocorid *Dindymus ventralis* Mayr, 1866, a species that is not documented in our *Heteroptera from Victoria* dataset needs confirmation. Five photographic specimens were only identified to family level; however, I am confident that, once identified to species, they will belong in different taxa. The dataset containing these records from the Flickr group *Heteroptera from Australia* is given in Table R1 (Supplementary materials).

4.2.4 *Flickr group: Heteroptera from the Iberian Peninsula*

A total of 382 photographs were curated from the group's pool. These were contributed by 44 photographers, which documented the occurrence of 160 heteropteran bug species in 149 different Iberian Peninsula localities. Records were attributable to 39 out of 67 (58%) of our working spatial units for the Iberian Peninsula. With a few exceptions, species identification was undertaken by the 45 members of the group. The following seven species had the largest number of photographic records: *Graphosoma lineatum italicum* (Müeller, 1766) (Figure 4.7) and *P. apterus* (Figure 3.7 and A3.7C), each with 19 records or 5.0% of the total; *C. m. marginatus* and *E. ornata* (12, 3.1%; Figures 4.4 and 4.3, respectively), and *Eurydema oleracea* (Linnaeus, 1758) (Figure 4.8), *S. aegyptius* (Figure 3.8) and *S. pandurus* (Figure 4.6), each with 9 records or 2.4% of the total. The dataset containing these records from the Flickr group *Heteroptera from the Iberian Peninsula* is given in Table R2 (Supplementary materials).

4.3 Heteropteran bug species

A total of 512 heteropteran bugs species were collected and identified in this work. Among these, 418 species were collected exclusively in the Iberian Peninsula, accounting for approx. 30% of the known heteropteran bug fauna of the bioregion.

Figure 4.2 A mating pair of the pentatomid species *Nezara viridula* (Linnaeus, 1758) photographed in Montgat (Barcelona, Catalonia, Spain). Source: original.

Figure 4.3 The pentatmid *Eurydema ornata* (Linnaeus, 1758) photographed in Collserola Natural Park (Barcelona, Catalonia, Spain). Source: original.

Figure 4.4 The Coreid *Coreus marginatus marginatus* (Linnaeus, 1758) photographed in Vitoria-Gasteiz (Álava, Basque Country, Spain). Source: original.

Figure 4.5 The pentatomid *Dolycoris baccarum* (Linnaeus, 1758) photographed in La Garrotxa Natural Park (Gerona, Catalonia, Spain). Source: original.

Figure 4.6 The lygaeid *Spilostethus pandurus* (Scopoli, 1763). Source: Stanislav Krejčík (Encyclopedia of Life).

The 95 species found exclusively in Victoria accounted for about 22% of the region's known heteropteran bugs. Only one species, the cosmopolitan *N. viridula* (Figure 4.2), was collected in both regions.

Representing about 14% of all species sampled, the seven most ubiquitous species were all herbivores: O*xycarenus lavaterae* (Fabricius, 1787) (Figure 4.9 and A3.7B), *E. oleracea* (Figure 4.8), *Aelia acuminata* (Linnaeus, 1758) (Figure 4.10), *Beosus maritimus* (Scopoli, 1763) (Figure A5.10), *D. baccarum* (Figure 4.5), *Mutusca brevicornis* (Dallas, 1852) (Figure A2.22) and *Rhopalus subrufus* (Gmelin, 1790) (Figure 4.11). The most ubiquitous predatory species were *Geocoris erythrocephalus* (Le Peletier and Serville, 1825) (Figure 4.12), *Orius laevigatus laevigatus* (Fieber, 1860) and *Nabis pseudoferus ibericus* Remane, 1962, accounting for under 4% of all species sampled. We also note here the presence among our identified species of the Mirids *Orthotylus caprai* Wagner, 1955 (Figure 4.13) and *Dicyphus maroccanus* Wagner, 1951, the former representing a first record for the Iberian Peninsula (Goula and Mata 2011a, Aukema et al. 2013), and the latter a second record for continental Europe. The heteropteran bug species identified here were members of 35 different families. As illustrated in Figure 4.14, the majority of species belonged to the cosmopolitan families Miridae (29.9 %), Pentatomidae (13.5%) and Rhyparochromidae (11.3%). Our surveys also allowed us to document the association of 74 heteropteran species with at least 58 different plant species. For the full dataset containing these new field records see Table R3 in the Supplementary materials.

Figure 4.7 A mating pair of the pentatomid species *Graphosoma lineatum italicum* (Müller, 1766) photographed in Cillaperlata (Burgos, Castille-Leon, Spain). Source: original.

Figure 4.8 The pentatomid *Eurydema oleracea* (Linnaeus, 1758). Source: Malcolm Storey (Encyclopedia of Life).

Figure 4.9 The oxycarenid *Oxycarenus lavaterae* (Fabricius, 1787) photographed in Els Poblets (Alicante, Valencia, Spain). Source: Katja Schulz (Encyclopedia of Life).

4.4 Faunistic catalogs and datasets

4.4.1 *Catalog of the Heteroptera from the Iberian Peninsula*

The Iberian Peninsula Heteroptera comprise 1,453 species. This total increases to 1,470 when the recognized subspecies are included. These 1,470 species and subspecies belong to 6 infraorders, 21 superfamilies, 46 families, 77 subfamilies, 89 tribes and 480 genera.

The distribution of Iberian Peninsula Heteroptera species among infraorders, superfamilies and families is shown in Table 4.1. The Cimicomorpha, with 805 species, accounts for approx. half of the Iberian Peninsula heteropterofauna, whereas Dipsocoromorpha, with only 3 species, accounts for only 0.2%. The superfamilies with the largest number of species are Miroidea, Lygaeoidea and Pentatomoidea, with 656, 269 and 176 species, respectively. With 557 species, the Mirids are the best represented family in the Iberian Peninsula, followed by Rhyparochromids (149 spp.) and Pentatomids (104 spp.). The Iberian fauna includes six monospecific families: Ceratocombidae, Hydrometridae, Aepophilidae, Pleidae, Ochteridae and Plataspidae. Likewise, the number of species per family or lower taxonomical levels (ie, subfamily or tribe) are given in Table R4 (Supplementary materials). With 206 and 181 species each, the Phylinae and Phylini are the most diversified subfamily and tribe, respectively. A total 55 subfamilies and tribes are represented by just one genus.

Our search for heteropteran bug citations in the entomological literature has

Figure 4.10 The pentatomid *Aelia acuminata* (Linnaeus, 1758) photographed in Cillaperlata (Burgos, Castille-Leon, Spain). Source: original.

Table 4.1 Summary of the known number of heteropteran bug species (Spp) by family (F) for the Iberian Peninsula bioregion. Families are grouped by superfamily (Sf), and these by infraorder (Io).

Io	Sf	F	Spp
Cimicomorpha			
	Cimicoidea		
		Anthocoridae	49
		Cimicidae	3
	Microphysoidea		
		Microphysidae	9
	Miroidea		
		Miridae	557
		Tingidae	99
	Naboidea		
		Nabidae	25
	Reduvioidea		
		Reduviidae	63
Dipsocoromorpha			
		Ceratocombidae	1
		Dipsocoridae	2
Gerromorpha			
Gerroidea			
		Gerridae	13
		Veliidae	9
Hebroidea			
		Hebridae	2
	Hydrometroidea		
		Hydrometridae	1
	Mesovelioidea		
		Mesoveliidae	2
Leptopodomorpha			
	Leptopodoidea		
		Leptopodidae	4
	Saldoidea		
		Aepophilidae	1
		Saldidae	22
Nepomorpha			
	Corixoidea		
		Corixidae	41
	Naucoroidea		
		Aphelocheiridae	2
		Naucoridae	3
	Nepoidea		
		Nepidae	2
	Notonectoidea		
		Notonectidae	10
		Pleidae	1
	Ochteroidea		
		Ochteridae	1
Pentatomomorpha			
	Aradoidea		
		Aradidae	27
	Coroidea		
		Alydidae	5
		Coreidae	36
		Rhopalidae	25
		Stenocephalidae	6
	Lygaeoidea		
		Artheneidae	3
		Berytidae	20
		Blissidae	3
		Cymidae	4
		Geocoridae	16
		Heterogastridae	6
		Lygaeidae	41
		Oxycarenidae	19
		Piesmatidae	8
		Rhyparochromidae	150
	Pentatomoidea		
		Acanthosomathidae	7
		Cydnidae	38
		Pentatomidae	104
		Plataspidae	1
		Scutelleridae	24
		Thyreocoridae	2
	Pyrrhocoroidea		
		Pyrrhocoridae	3
Total			**1470**

Based on a summary of all sourced used in this work.

Figure 4.11 The rhopalid *Rhopalus subrufus* (Gmelin, 1790). Source: Miroslav Deml (Encyclopedia of Life).

Figure 4.12 The geocorid *Geocoris erythrocephalus* (Le Peletier & Serville, 1825). Source: Wikimedia Commons.

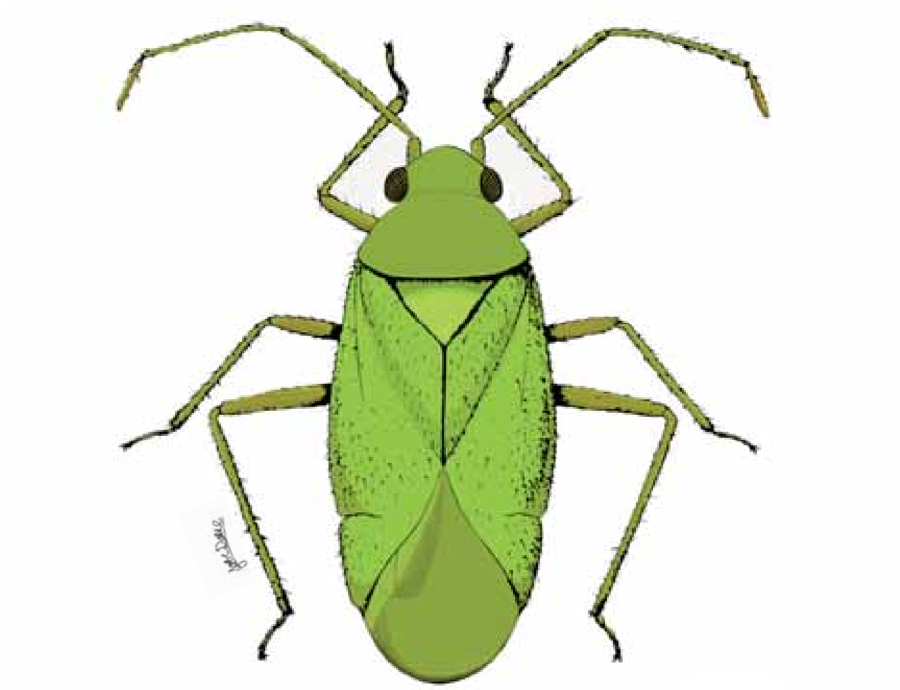

Figure 4.13 Among the species collected and identified for this work was the mirid *Orthotylus caprai* Wagner, 1955 which represented a first record for the Iberian Peninsula bioregion (Goula and Mata 2011a, Aukema et al. 2013). Source: José Doble.

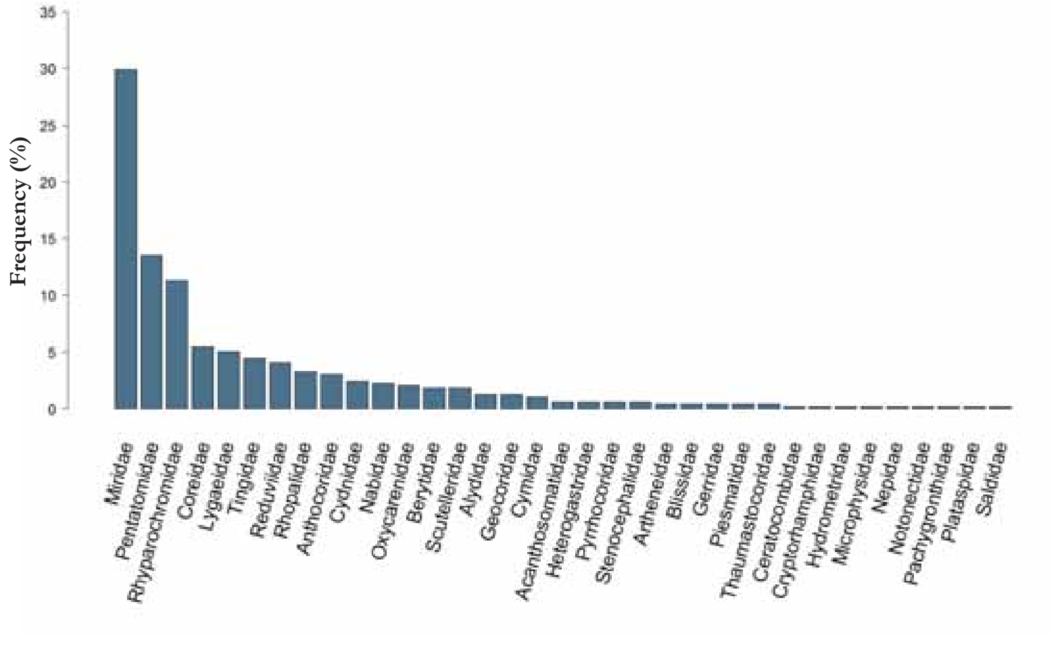

Figure 4.14 Rank by frequency plot for the 35 heteropteran bug families collected during the present work.

yielded thus far 14,082 records. These were found among 439 works, spanning the timeframe between the years 1814 and 2013. The species with the largest number of documented records assignable to one of our working 67 Iberian Peninsula spatial units were *Nepa cinerea* (Figure A3.10A), with 183 records, *Naucoris maculatus* (129; spp. *conspersus* 16, spp. *maculatus* 113) and *Dictyla echii* (125). Our dataset indicates that 169 species in the bioregion are documented exclusively by the 'Catalogue of the Heteroptera of the Palaearctic Region' (Aukema and Rieger 1995, 1996, 1999, 2001, 2006, Aukema et al. 2013). Since, the occurrence of species in the abovementioned work are, in general, documented at the country level, it was not possible in most cases to assign these species to any of our spatial units. The spatial units with the largest number of assignable records were the Spanish provinces of Barcelona (1,490), Madrid (524) and Zaragoza (491), whereas the Portuguese districts of Castelo Branco (17), Portalegre (20) and Braga (28) were the spatial units with the smallest number of records.

Data stemming from the biodiversity web resource Biodiversidad Virtual allowed us to document the occurrence of 603 heteropteran bug species in 2,573 different Iberian Peninsula localities. These localities belonged to 63 (94%) of our working spatial units for the Iberian Peninsula. Likewise, through the Flickr group *Heteroptera from the Iberian Peninsula* we documented the occurrence of 160 species in 149 localities, which were assignable to 39 spatial units. As previously reported in *Heteropteran bug specimens* above, the present work contributed 3,572 new Iberian Peninsula field records of 418 different heteropteran bug species.

Publication of the *Catalog of the Heteroptera from the Iberian Peninsula* has been agreed with A. Melic, chief editor of the 'Sociedad Entomólogica Aragonesa'. The multi-authored catalog will be published family by family as part of the society's 'Online Monographic Series'. The Rhyparochromidae, the first family of the series, is presented in Appendix I.

4.4.2 *Heteroptera from El Maresme*

El Maresme heteropterofauna comprises 323 species, belonging in 33 families. With 90 species, the Miridae are the best represented family, followed by Pentatomidae (49 spp.) and Rhyparochromidae (24 spp.). Our search for records of heteropteran bugs in El Maresme has yielded 1,860 records. The large majority of these (approx. 58%) were new field records provided by the 2011 insect survey and the unpublished specimens labeled as 'other material studies'. Bibliographical references accounted for 40.7% of records, while photographic records accounted for only 1.9%. Accounting for 3.1% of all records, *E. oleracea* (Figure 4.8) may be considered the most ubiquitous species in the region. This species was followed in frequency of records by *Nysius graminicola graminicola* (Kolenati, 1845) and *O. lavaterae* (Figure 4.9), accounting each for 3.0% of all records, *Macrolophus melanotoma* (A. Costa, 1853) (2.5%), *B. maritimus* (2.2%; Figure A5.10), *O. l. laevigatus* (2.0%) and *Closterotomus trivialis* (A. Costa, 1853) (1.9%; Figure 4.15).

The municipality with the highest number of documented species was Calella (169 spp.), followed by Mataró (103 spp.) and Montgat (69 spp.). On the other hand, the municipality with the least number of species were Vilassar de Mar (18 spp.), Premià de Mar (20 spp.) and Premià de Dalt (22 spp.). Interestingly, many of the

Figure 4.15 The mirid *Closterotomus trivialis* (A. Costa, 1853) photographed in Teià (Barcelona, Catalonia, Spain). Source: original.

taxa that contributed to Calella and Mataró being species-rich municipalities were documented in bibliographical references that were at least 100 years old (eg, Salvañá 1870, Bolívar and Chicote 1879). A considerable number of species, genera and families documented in these municipalities through these older works have not been re-recorded in El Maresme thereafter. This is the case, for example, of the pentatomomorphan families Acanthosomatidae, Aradidae, Cymidae, Plataspidae and Stenocephalidae. We also note the new field record of *Piesma maculatum* (Laporte, 1833 , which comes approx. 100 years after it was first documented in Sánchez (1920). Finally, we highlight the new field record of *Ischnodemus sabuleti* (Fallén, 1826), which also represents the first record in El Maresme of the family Blissidae.

4.4.3 Pyrrhocoridae from the Iberian Peninsula

The Iberian Peninsula Pyrrhocoridae comprise two species: *Pyrrhocoris apterus* (Linnaeus, 1758) (Figure 3.7 and A3.7C) and *Scantius aegyptius* (Linnaeus, 1758) (Figure 3.8). The latter species being represented in the bioregion by two subspecies: *S. aegyptius aegyptius* (Linnaeus, 1758) and *S. aegyptius rossii* Carapezza, Kerzhner & Rieger, 1999.

Our search of the entomological literatures yielded 72 and 51 records of *P. apterus* and *S. aegyptius*, respectively. These records were found among 45 different papers, the oldest dating back as far as 1877. Bibliographical records placed *P. apterus* in 42% of our working spatial units and *S. aegyptius* in 39% of them. The largest

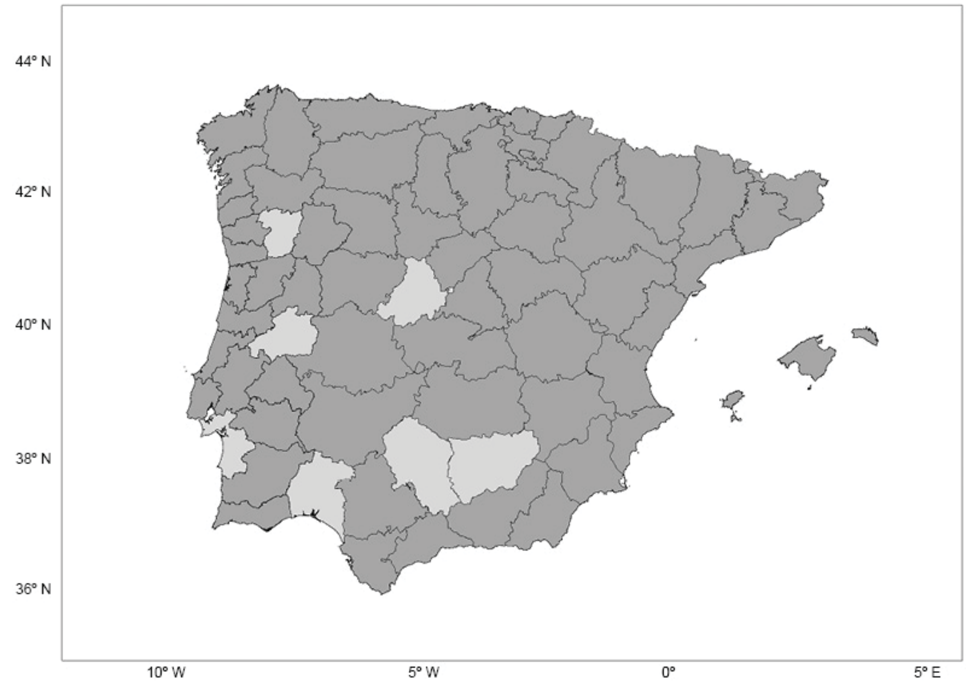

Figure 4.16 Distribution of *Pyrrhocoris apterus* in the Iberian Peninsula bioregion. Dark grey: Species present as documented by bibliographic, field or photographic records. Light grey: Species absent or not yet recorded. The names of the spatial units are given in Figure 3.5. Source: original.

amount of these records belonged to the Spanish province of Barcelona (11%) and the Portuguese district of Faro (8%). A total of 12 spatial units were documented exclusively by one reference in the literature. To these distribution data we added records stemming from 284 curated photographs of *P. apterus* (232) and *S. aegyptius* (51). Over 92% of these photographic records were provided by *Biodiversidad Virtual*. These photographic records included localities within 48 of our spatial units (73%), with the largest amount of records belonging to the Spanish provinces of Barcelona (12%) and Madrid (9%). Records stemming from the literature and photographic web resources were further complemented with 116 new field records from *P. apterus* (110) and *S. aegyptius* (6). Among this field material were the first published records of *P. apterus* for the Portuguese districts of Aveiro, Braga, Santarém and Viana do Castelo, and for the Spanish provinces of Álava, Almería, Asturias, La Rioja and Vizcaya.

These new combined data, indicates that *P. apterus* occurs in 60 of our Iberian Peninsula spatial units, while *S. aegyptius* occurs only in 31. Since the surface area of each spatial unit is well known (Instituto Geográfico Nacional 2013, Instituto Nacional de Estatística 2013), we were able to estimate that *P. apterus* and *S. aegyptius* are roughly distributed through 90% and 51%, respectively, of the territory of the Iberian Peninsula bioregion. The known distributions of *P. apterus* is illustrated in Figure 4.16, while that of *S. aegyptius* is given in Figure 4.17. The datasets containing the bibliographical records and the photographic records are given in Table R5 and R6, respectively (Supplementary materials).

A manuscript titled “Pyrrhocoridae from the Iberian Peninsula” by L. Mata, J.M. Grosso-Silva and M. Goula has been submitted to *Heteropterus Revista de Entomología*, and is presently under evaluation by the editors of the journal.

4.4.4 *Heteroptera from Victoria*

The known heteropterofauna from Victoria, Australia comprise 437 species. This number increases to 438 when the only non-nominotypical subspecies *Micronecta annae tasmanica* Wróblewski, 1977 is included. These species belong to 5 infraorders, 17 superfamilies, 48 families, 68 subfamilies, 61 tribes and 277 genera. With 103 species, Pentatomidae is the best represented family, followed by Reduviidae (58 spp.) and Miridae (50 spp.). Species among these three families represent almost 50% of the total heteropteran bug fauna of the region. More than half of the families are represented by less than four species. Belonging all to the Pentatomidae, the taxa Pentatominae (90 spp.), Rhynchocorini (21 spp.) and *Poecilometis* (9 spp.) are the most diversified subfamily, tribe and genus, respectively.

The first volume of the “Zoological Catalogue of Australia” (Cassis and Gross 1995) documented the occurrence in Victoria of 168 heteropteran bug species, while the second volume (Cassis and Gross 2002) documented 249 species. Our search of the entomological literature posterior to this groundwork yielded 12 papers, which

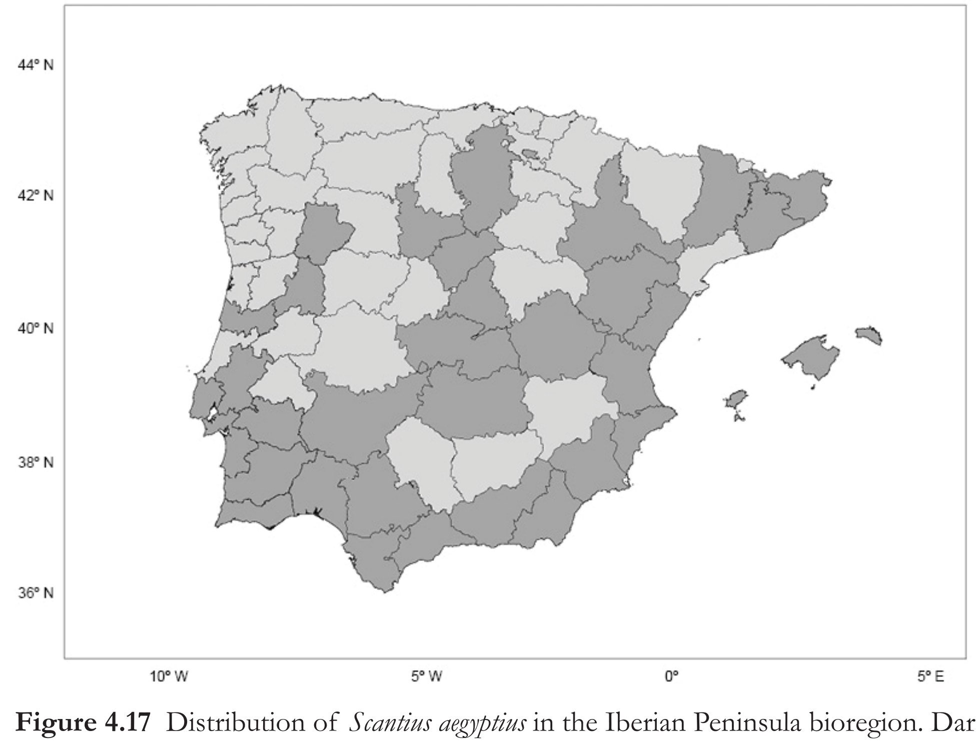

Figure 4.17 Distribution of *Scantius aegyptius* in the Iberian Peninsula bioregion. Dark grey: Species present as documented by bibliographic, field or photographic records. Light grey: Species absent or not yet recorded. The names of the spatial units are given in Figure 3.5. Source: original.

through new synonymies, combinations and description of species brought up the total number of known species to 435. In total, these 14 works documented at least 449 records of heteropteran bugs assignable to Victoria.

Our 2012 insect survey conducted in south-east Melbourne yielded 7,659 heteropteran bug specimens (see *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*). These translated into 746 new field records for Victoria. Specimens belonged to at least 95 heteropteran bug taxa. Thus far, 38 of these taxa have been identified to species level, including the following three that we report here for the first time in the Victoria region: *Crompus oculatus* Stål, 1874 (Lygaeidae), *Koscocrompus obscurus* Scudder, 1958 (Lygaeidae) and *Melanacanthus scutellaris* (Dallas, 1852) (Alydidae). Of the remaining 57 taxa, 17 have been identified to genus and the rest to tribe, subfamily or family. In all cases we are confident that these 57 taxa correspond to separate species and that at least some of them will turn out to be new to Victoria, to Australia or to science.

For the dataset containing the 438 species and subspecies see Table R7 in the Supplementary materials. This table includes the taxonomical affiliations of each species and the bibliographical references where the occurrence in Victoria of the species is documented. When appropriate the synonymies and new name combinations are also given.

4.5 New diagnostic dichotomous keys

A total of 157 new diagnostic dichotomous keys were developed for the present work. These included two general keys to family level: *Key to the families of Heteroptera from Victoria* (Appendix II) and *Key to the families of Heteroptera from the Iberian Peninsula* (Appendix III). Also included among our developed keys were 10 family to genera/species keys, eight family to subfamily keys, one family to tribe key, namely our *Key to the tribes of Miridae from the Iberian Peninsula* (Appendix IV), three subfamily to tribe keys, 15 subfamily to genera/species keys, including our *Key to the genera of Rhyparochrominae from the Iberian Peninsula* (Appendix V), 16 tribe to genera/species keys, and 102 genus to species keys. An example of this latter group was our *Key to the species of Deraeocoris from the Iberian Peninsula* (Appendix VI).

4.6 Case studies

4.6.1 The insect biodiversity benefits of novel grassland ecosystems in urban green spaces

Our insect survey yielded 6,628 heteropteran bug specimens, belonging to 76 different species (67 herbivores and 9 predators) in 22 families. Of these species, 42 and 61 were observed in woodland and grassland, respectively. The most frequent and abundant herbivore was the alyidid *M. brevicornis* (Figure A2.22), representing almost 30% of all specimens, while the most frequent and abundant predatory species was

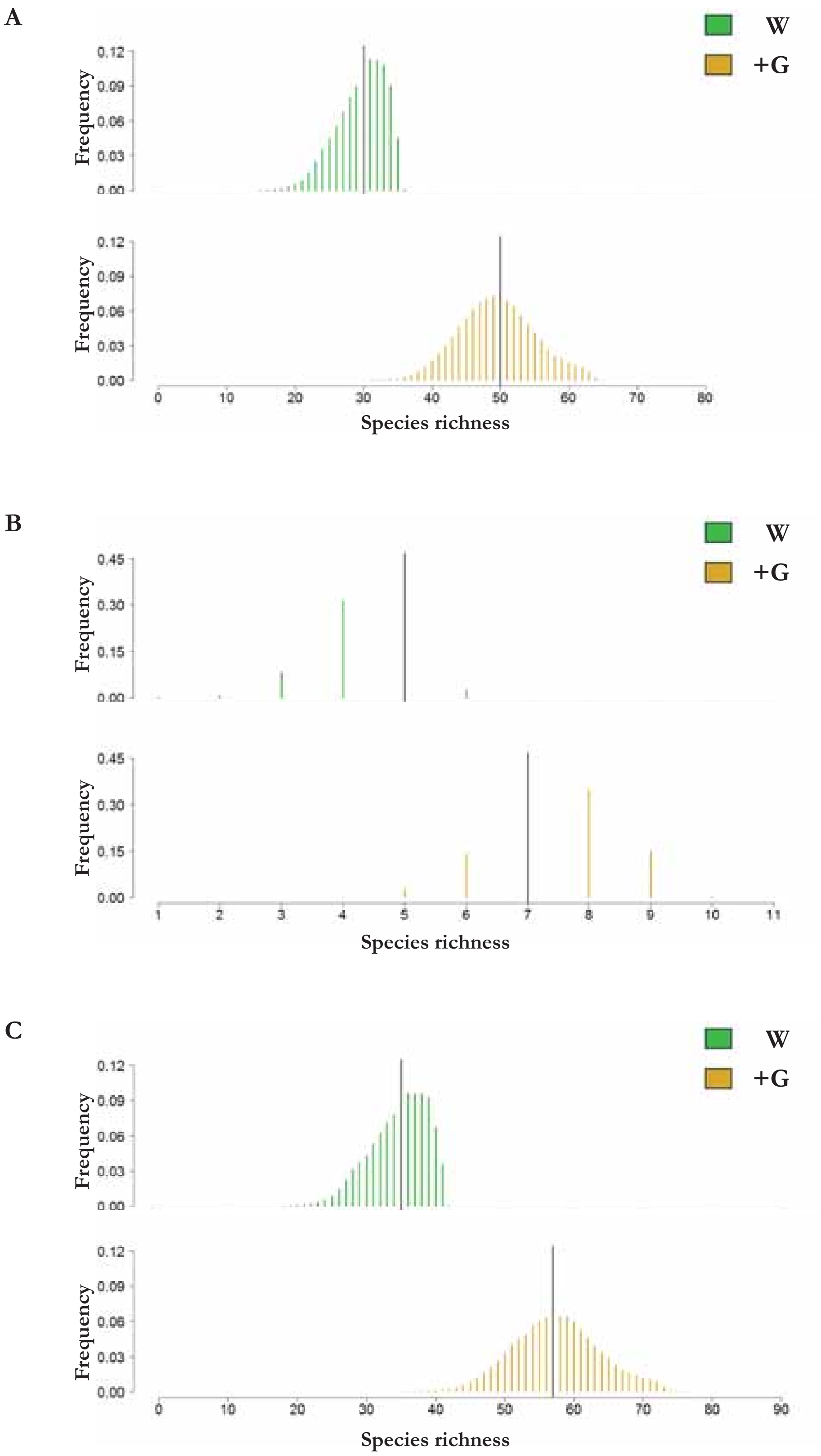

Figure 4.18 Comparisons of the posterior distributions of heteropteran bug species richness under the 'woodland' (**W**) and 'woodland plus grassland' (**+G**) models. **A.** Herbivorous guilds. **B.** Predatory guild. **C.** Whole community.

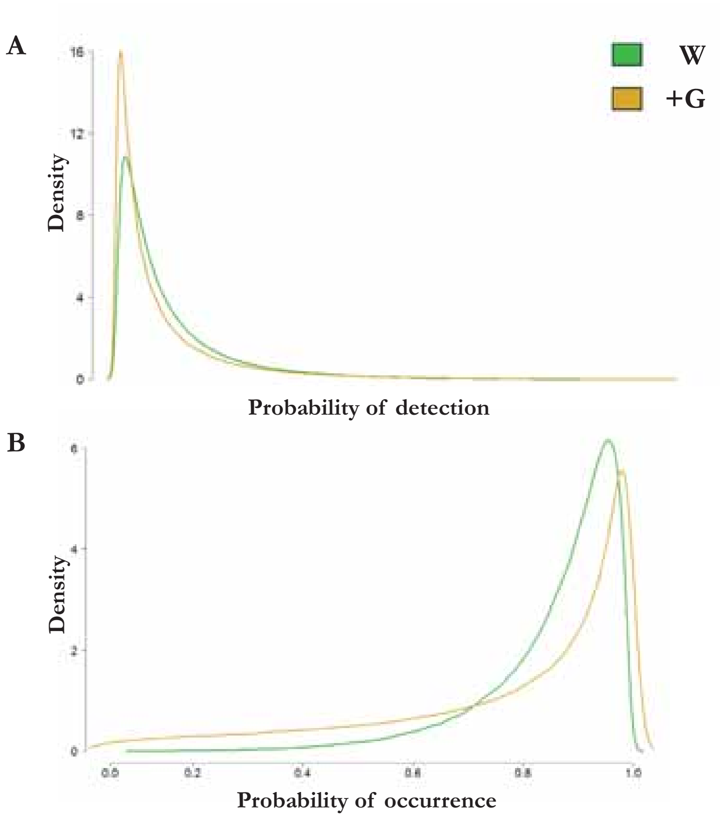

Figure 4.19 Comparisons of the distributions of probabilities of occurrence and detection of heteropteran bugs based on the estimates of the 'woodland' (W) and 'woodland plus grassland' (+G) models.

the nabid *Nabis kinbergii* (Figure A2.17), representing approx. 4% of all specimens.

Our woodland models estimated 35 heteropteran bug species (95% Credible interval: 26-41) with 30 (23-35) and 5 (3-6) herbivorous and predatory species, respectively, while the model combining the woodland and grassland data estimated 60 species (49-70), with 52 (43-60) herbivores and 8 (6-10) predators (Figure 4.18A-C and Table 4.2). Mean estimates for the probabilities of occurrence were high both in the woodland (0.866; CI: 0.644 - 0.991) and woodland plus grassland (0.914; CI: 0.730 – 0.998) models (Figure 4.19B and Table 4.2). On the other hand, detection was low, varying from 0.036 to 0.098 (mean=0.061) in woodland, and from 0.034 to 0.061 (mean=0.041) in woodland plus grassland (Figure 4.19A and Table 4.2). As illustrated in Figure 4.20A-C, woodland plus grassland species richness estimates of the whole heteropteran community, as well as those of the herbivorous and predatory guilds, showed an increasing trend along the vegetation density gradient. The mean community-level effect of vegetation density (VD) on the occurrence probability of heteropteran bugs was positive, and the posterior credible interval for this community hyperparameter contained only positive values (Figure 4.21 and Table 4.2). The probability of vegetation density having a positive effect on the occurrence probability of heteropteran bugs was 0.928. This probability was estimated as the

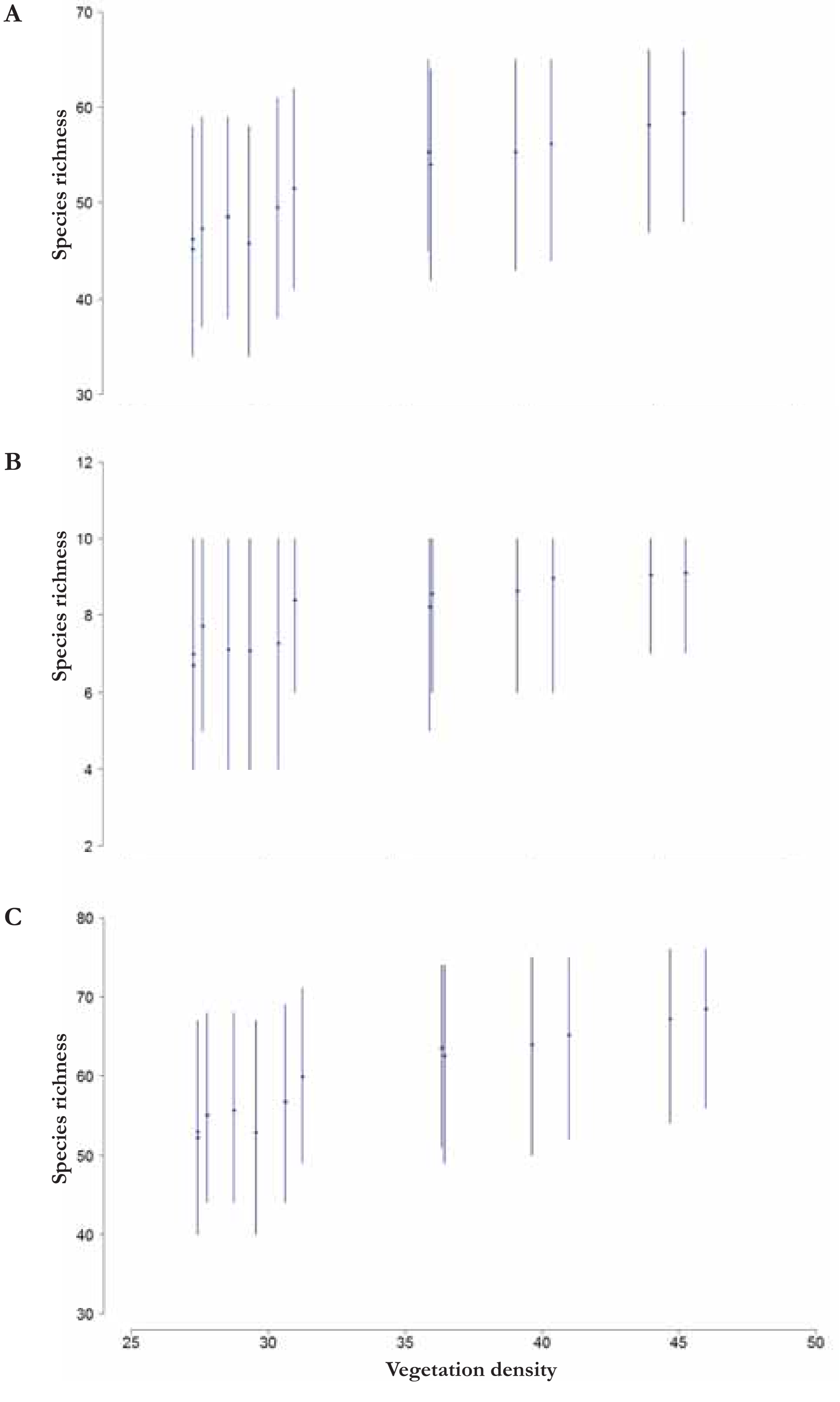

Figure 4.20 Relationships between species richness and vegetation density. **A.** Herbivorous guild. **B.** Predatory guild. **C.** Whole community. The solid dots indicates means and the vertical bars 95% credible interval.

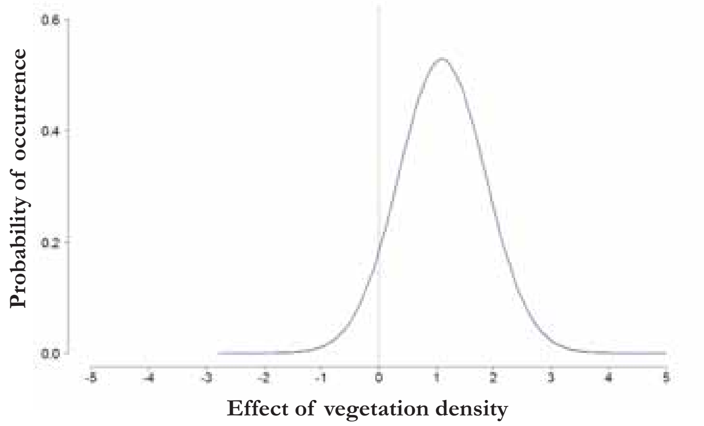

Figure 4.21 Mean community-level effect of vegetation density on the occurrence probability of heteropteran bugs (mean=1.102; CI: 0.054, 2.938).

Table 4.2 Community-level summaries of the hyperparameters for occurrence, detection, vegetation density covariate and species richness.

Community-level hyper-parameters	Woodland model				Full model			
	Mean	SD	2.5	97.5	Mean	SD	2.5	97.5
Occurrence								
Mean	0.866	0.093	0.644	0.991	0.914	0.072	0.730	0.998
SD	0.737	0.731	0.644	0.991	0.930	0.704	0.772	0.990
Detection								
Mean	0.061	0.016	0.036	0.098	0.041	0.009	0.034	0.061
SD	0.765	0.543	0.707	0.826	0.806	0.536	0.763	0.849
Vegetation density								
Mean	—	—	—	—	1.102	0.756	0.054	2.938
SD	—	—	—	—	0.909	0.689	0.226	2.776
Species richness								
Herbivores	30.1	3.5	22.5	35.3	51.7	4.5	42.5	60.4
Predators	5.0	0.7	3.4	5.9	7.9	0.9	6.2	9.5
Total	35.1	4.1	26.3	41.2	59.7	5.1	49.3	69.5

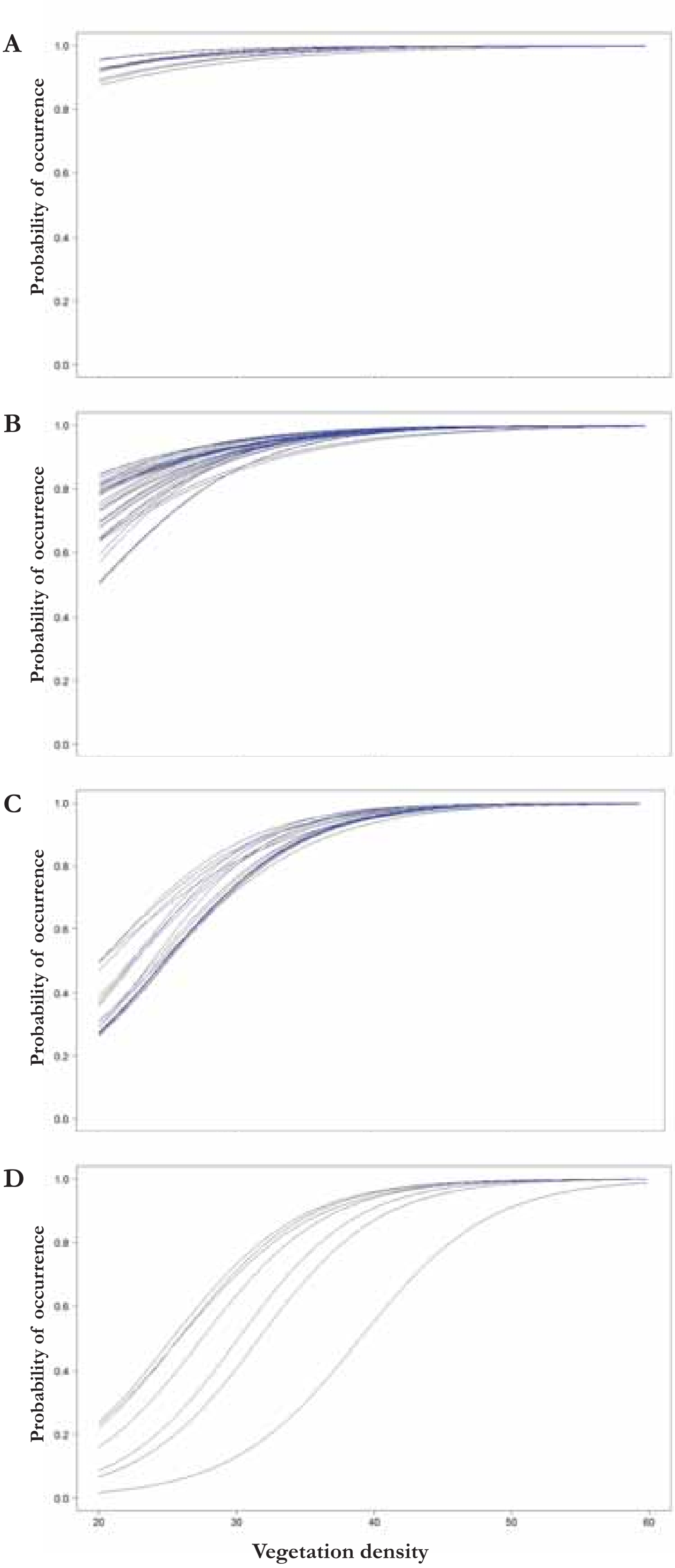

Figure 4.22 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and vegetation density. **A.** Species showing no response. **B.** Species showing a slight positive response. **C.** Species showing a moderate positive response. **D.** Species showing a large positive response.

area under the curve of the mean community-level effect hyperparameter.

Means, standard deviations and credible intervals for the 76 species-specific occurrence and detection probabilities, as well as the species-specific effects of VD on occupancy, are presented in Table R8 (Supplementary materials). Mean detection probabilities varied considerably among species (0.021 – 0.757), with the most abundant species showing the highest probabilities of detection. Mean probabilities of occurrence also varied considerably among species, ranging from 0.324 to 0.978.

We used the model's posterior distribution estimates to predict heteropteran bug occurrence probability for 500 values within a reasonable range of the vegetation density gradient (20-60). These predictions were then used to illustrate the species-specific relationships between occupancy and vegetation density (Figures 4.22A-D and Table R8). Vegetation density (VD) had a mix of different effects on the species-specific mean probabilities of occurrence. Seven and two, herbivores and predators, respectively, showed a large increase (greater than 4-fold change) in occurrence probability as VD increased over the range of characterized golf courses (Figure 4.22D). Among the herbivores, 17 species showed a moderate increase (greater than 2 to 4-fold change), while only one predatory species (the geocorid *Stylogeocoris biroi* Montandon, 1913) showed the same moderate response (Figure 4.22C). The number of herbivorous species showing a slight response (a change in occurrence probability greater than 1.15 but smaller than 2) was 32, while five predatory species showed this same slight response (Figure 4.22B). Finally, 12 herbivores and two predators showed no response (a change larger than 0.85 but smaller than 1.15) in occurrence probability from the less to the most densely-vegetated golf-course in our study area (Figure 4.22A).

4.6.2 *The effect of landscape complexity on vineyard biodiversity*

The insect survey yielded 149 heteropteran bug species: 119 herbivores (80%) and 30 predators (20%). Of the 910 specimens detected, 47 (5%) and 863 (95.5%) were sampled by the pitfall (PF) and flight intercept (FI) traps, respectively. The most frequent and abundant herbivores were the oxycarenid *O. lavaterae* (Figure 4.9) and the rhyparochromid *B. maritimus* (Figure A5.10), together representing approx. one fourth of all herbivores detected. On the other hand, the most frequent and abundant predatory species were the geocorid *G. erythrocephalus* (Figure 4.12) and the reduviid *Rhynocoris cuspidatus* Ribaut, 1921 (Figure 4.23), accounting for approx. 21% of all specimens in the predatory guild.

Of the total 119 herbivorous species observed during the survey, 70 were observed in Castellet i La Gornal (CG) and 96 in Avinyó Nou (AN), while of the total 30 predatory species, 18 and 22 were observed in CG and AN, respectively. As illustrated in Figure 4.24A, our unconditional model for the herbivorous guild estimated a mean of 48 species in CG (95% Credible interval: 32-68) and a mean of 70 in AN (55-84). The predatory guild model estimated a mean of 13 (6-24) and 11 (7-17) species in CG and AN, respectively (Figure 4.24B).

Mean probability of occurrence of herbivorous species was higher (0.597, CI: 0.447 - 0.736) in AN than in CG (0.378, CI: 0.209 – 0.592) (Figure 4.25A), while the mean probability of occurrence of predatory species between CG (0.450, CI: 0.175 –

0.902) and AN (0.363, CI: 0.185 – 0.602) was similar (Figure 4.25B). In general, mean probabilities of detection were low. For both herbivorous and predatory species, the FI trap showed higher detection probabilities when compared to PF, while the probability of detecting species in CG and AN was similar. The mean community-level effect of proportion of natural habitat (PNH) on the occurrence probabilities

Figure 4.23 An assassin-bug of the species *Rhynocoris cuspidatus* Ribaut, 1921 (Reduviidae) photographed in Monfragüe National Park (Extremadura, Spain). Source: original.

Table 4.3 Community-level summaries of the hyperparameters for occurrence, detection and the covarite effect of proportion of natural habitat.

Community-level hyper-parameters	Herbivores				Predatros			
	Mean	SD	2.5	97.5	Mean	SD	2.50%	97.5
Occurrence								
CG	0.378	0.101	0.002	0.006	0.450	0.195	0.175	0.009
AN	0.597	0.074	0.004	0.007	0.363	0.106	0.002	0.006
Detection								
FICG	0.027	0.008	0.014	0.050	0.020	0.009	0.008	0.044
FIAN	0.030	0.006	0.020	0.044	0.044	0.018	0.017	0.086
PFCG	0.001	0.001	0.000	0.002	0.002	0.002	0.000	0.007
PFAN	0.000	0.000	0.000	0.001	0.003	0.002	0.000	0.009
Covariate effect								
PNH	0.684	0.208	0.331	1.150	0.523	0.468	-0.327	1.549

CG. Castellet i La Gornal; AN: Avinyó Nou; FI: Flight intercept trap; PF: Pitfall trap; PNH: Proportion of natural habitat

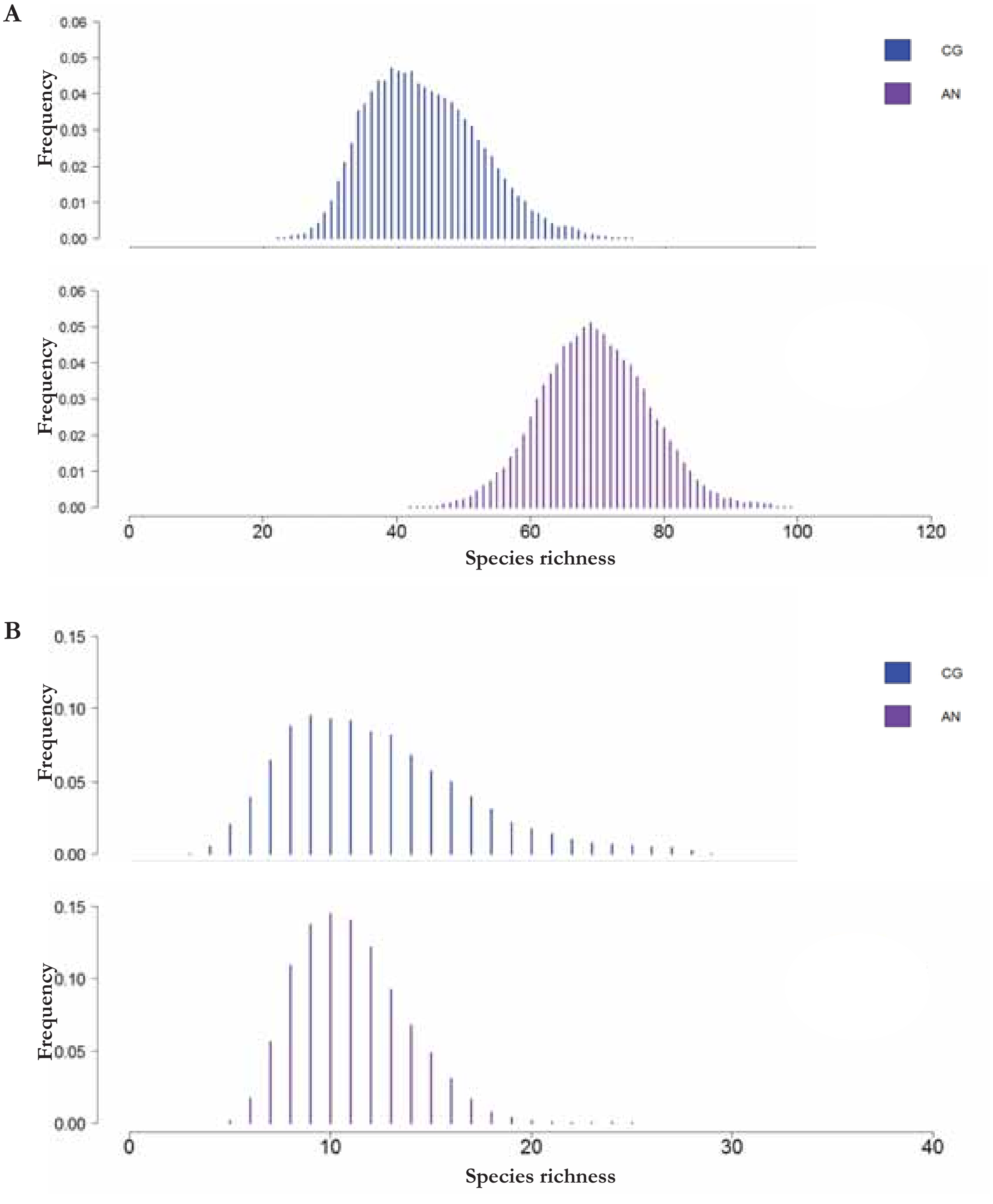

Figure 4.24 Comparisons of the posterior distributions of heteropteran bug species richness for Castellet i La Gornal (**CG**) and Avinyó Nou (**AN**). **A.** Herbivorous guild. **B.** Predatory guild.

of both herbivorous and predatory species was positive. The credible interval of the herbivorous guild hyper-parameter contained only positive values, while the one of the predatory guild had both negative and positive values. Table 4.3 gives the mean, standard deviation, and credible interval values for the occurrence and detection hyperparameters, as well as the guild-level effects on occurrence.

The mean, standard deviation, and credible intervals for the 115 herbivorous and 30 predatory species-specific occurrence and detection probabilities estimated using the FI data, as well as the species-specific effects of PNH on occurrence, are presented in Tables R9 (Herbivores) and R10 (Predators) in the Supplementary materials. Mean detection probabilities varied considerably among species (0.019 – 0.220), while remaining similar between guilds. The most abundant herbivores and predators, as described above, showed the highest probabilities of detection. On the other hand, mean probabilities of occurrence varied widely among species and between guilds, ranging from 0.378 to 0.730 among herbivores, and from 0.223 to 0.348 among predatory species.

We used the model's posterior distribution estimates to predict heteropteran bug occurrence probability for 500 values within the whole range of the proportion of

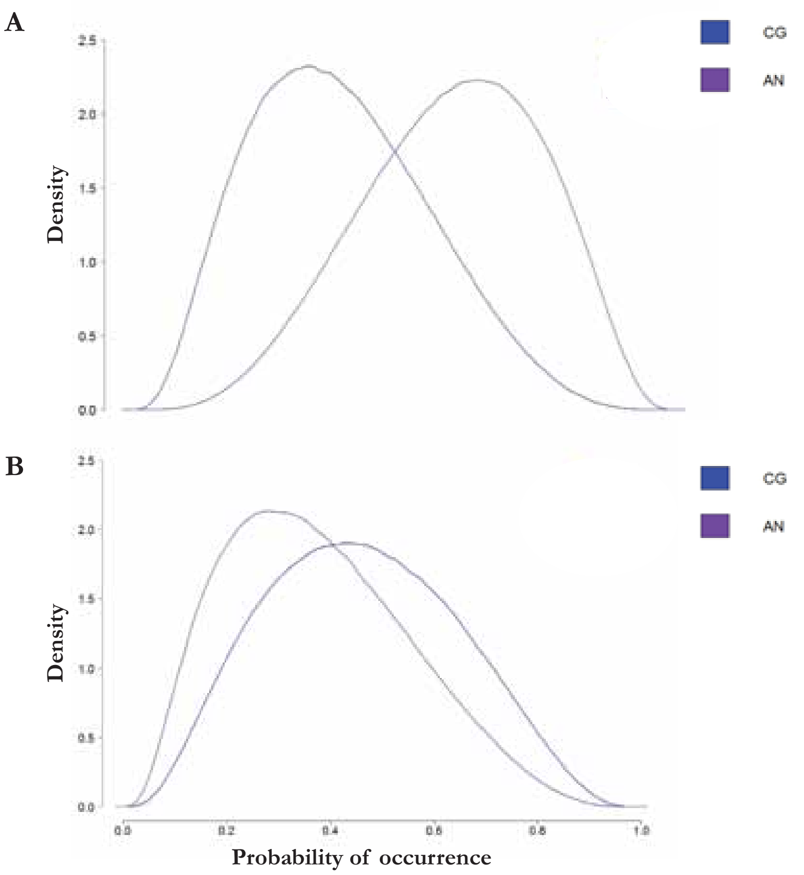

Figure 4.25 Comparisons of the distributions of probabilities of occurrence for Castellet i La Gornal (**CG**) and Avinyó Nou (**AN**). **A.** Herbivorous guild. **B.** Predatory guild.

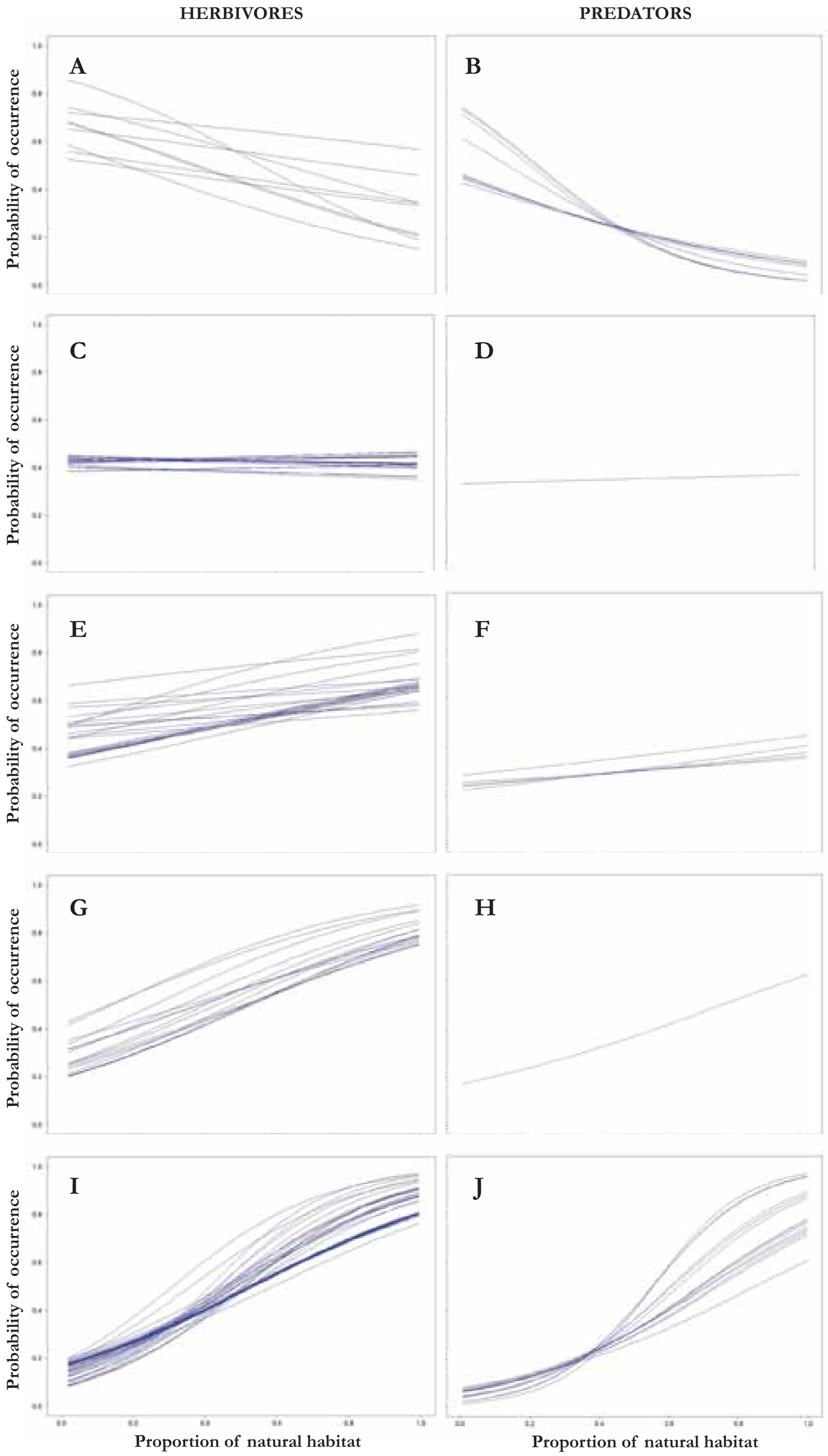

Figure 4.26 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and proportion of natural habitat. **A+B.** Species showing a negative response. **C+D**. Species showing no response. **E+F.** Species showing a slight positive response **G+H.** Species showing a moderate positive response. **I+J.** Species showing a large positive response.

natural habitat (PNH) gradient. These predictions were then used to illustrate the species-specific relationships between occupancy and PNH (Figures 4.26A-J and Tables R8 and R9). Proportion of natural habitat had a mix of different effects on the herbivorous and predatory species-specific mean probabilities of occurrence. A total of 45 and 14, herbivores and predators, respectively, showed a large increase (greater than 4-fold change) in occurrence probability as PNH increased over the range of characterized vineyard sites (Figures 4.26I-J). Among the herbivores, 17 species showed a moderate increase (greater than 2 to 4-fold change) (Figure 4.26G), while only one predatory species (the geocorid *G. erythrocephalus*) showed the same moderate response (Figure 4.26H). A total of 24 and 5, herbivores and predators, respectively, showed a slight increase (greater than 1.15 to 2.00 fold change) in mean estimates of occupancy (Figures 4.26E-F). The number of herbivorous species showing no response (a change in occurrence probability greater than 0.85 but smaller than 1.15) was 20 (Figure 4.26C), while only one predatory species (the anthocorid *Lyctocoris campestris* (Fabricius, 1794)) showed no response (Figure 4.26D). Finally, nine species in each guild showed a decrease (smaller than 0.85 fold change) in occurrence probability from the vineyard with the smallest to largest amount of PNH (Figures 4.26A-B).

4.6.3 *Effects of urbanization on occupancy and species richness*

Our insect survey yielded 142 heteropteran bug species: 112 herbivores (79%) and 30 predators (21%). The most frequently collected herbivores were the oxycarenid *O. lavaterae* (Figure 4.9), the lygaeid *N. g. graminicola* and the pentatomid *E. oleracea* (Figure 4.8), together representing approx. one sixth of all detections. On the other hand, the most frequently collected predatory species were the anthocorid *O. l. laevigatus*, the mirid *M. melanotoma* and the geocorid *G. erythrocephalus* (Figure 4.12), together accounting for approx. one twelfth of all detections.

Our models estimated 108 heteropteran bug species (95% Credible interval: 93-121) with 85 (73-96) and 23 (19-26) herbivorous and predatory species, respectively (Figures 4.27A-C and Table 4.4). Mean probability of occurrence was high (0.863) ranging from 0.712 to 0.970 (Figure 4.28B and Table 4.4), whereas detection was low (0.047) varying from 0.036 to 0.061 (Figure 4.28A and Table 4.4). As illustrated in Figures 4.29A-C, city-level species richness estimates of the whole heteropteran community, as well as those of the herbivorous and predatory guilds, showed a decreasing trend along the urbanization gradient. The mean community-level effect of urbanization (URB) on the occurrence probability of heteropteran bugs was negative, and the posterior credible interval for this community hyperparameter contained only negative values (Figure 4.30 and Table 4.4).

Means, standard deviations and credible intervals for the 142 species-specific occurrence and detection probabilities, as well as the species-specific effects of URB on occupancy, are presented in Table R11 (Supplementary materials). Mean detection probabilities varied considerably among species (0.018 – 0.517). As expected, the most frequently collected species, as described above, showed the highest probabilities of detection. Mean probabilities of occurrence varied among species from 0.614 to 0.979.

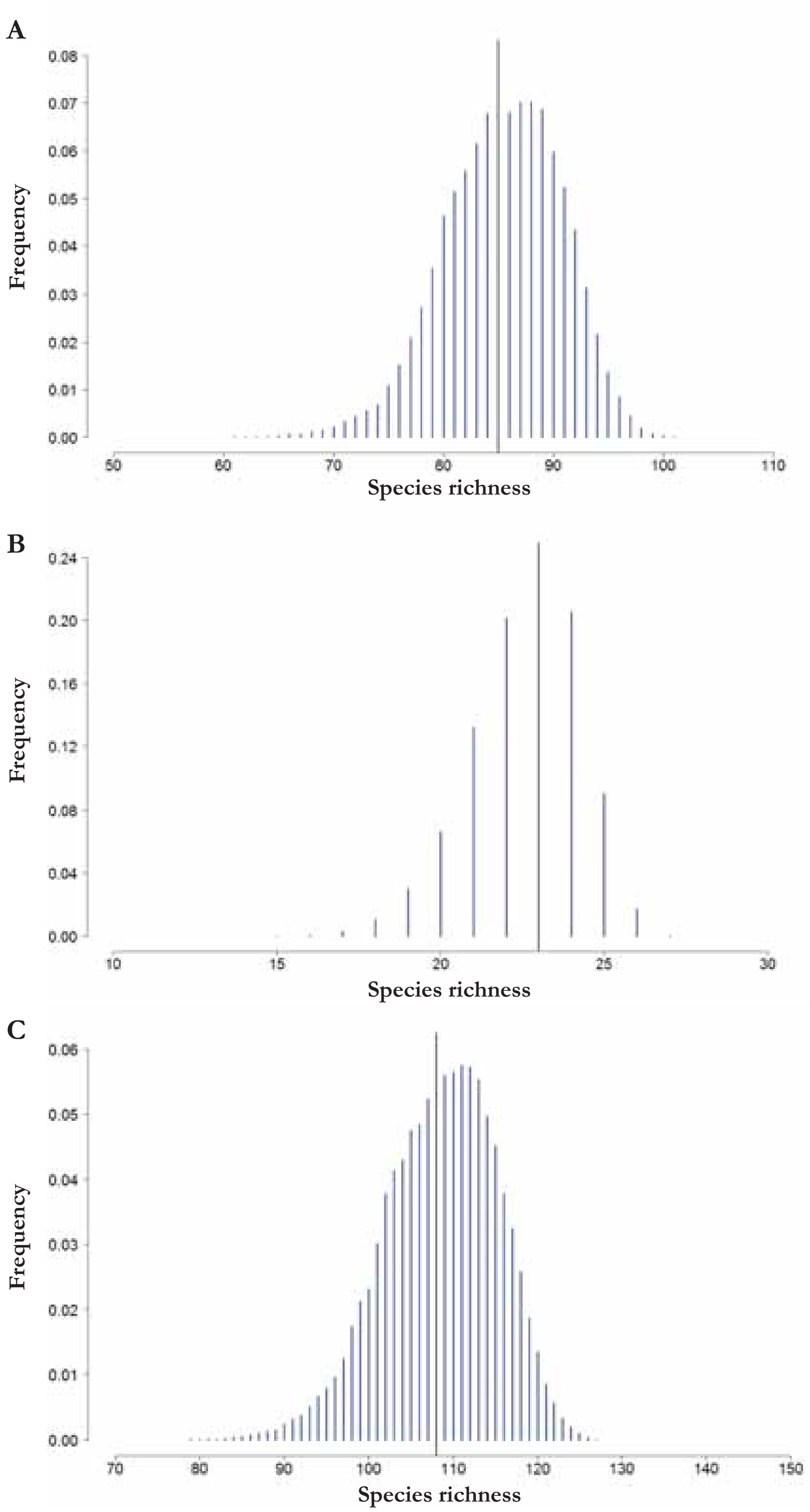

Figure 4.27 Posterior distributions of heteropteran bug species richness for El Maresme shire. **A.** Herbivorous guild. **B.** Predatory guild. **C.** Whole community.

Table 4.4 Community-level summaries of the hyperparameters for occurrence, detection, urbanization covariate and species richness.

Community-level hyper-parameters		Mean	SD	2.5	97.5
Occurrence					
	Mean	0.863	0.067	0.712	0.97
	SD	0.825	0.634	0.653	0.942
Detection					
	Mean	0.047	0.007	0.036	0.061
	SD	0.767	0.523	0.734	0.800
Urbanization					
	Mean	-1.154	0.371	-2.013	-0.576
	SD	0.927	0.42	0.279	1.938
Species richness					
	Herbivores	85.2	5.7	72.7	95.4
	Predators	22.9	1.7	19.1	25.9
	Total	108.1	7.1	92.6	120.8

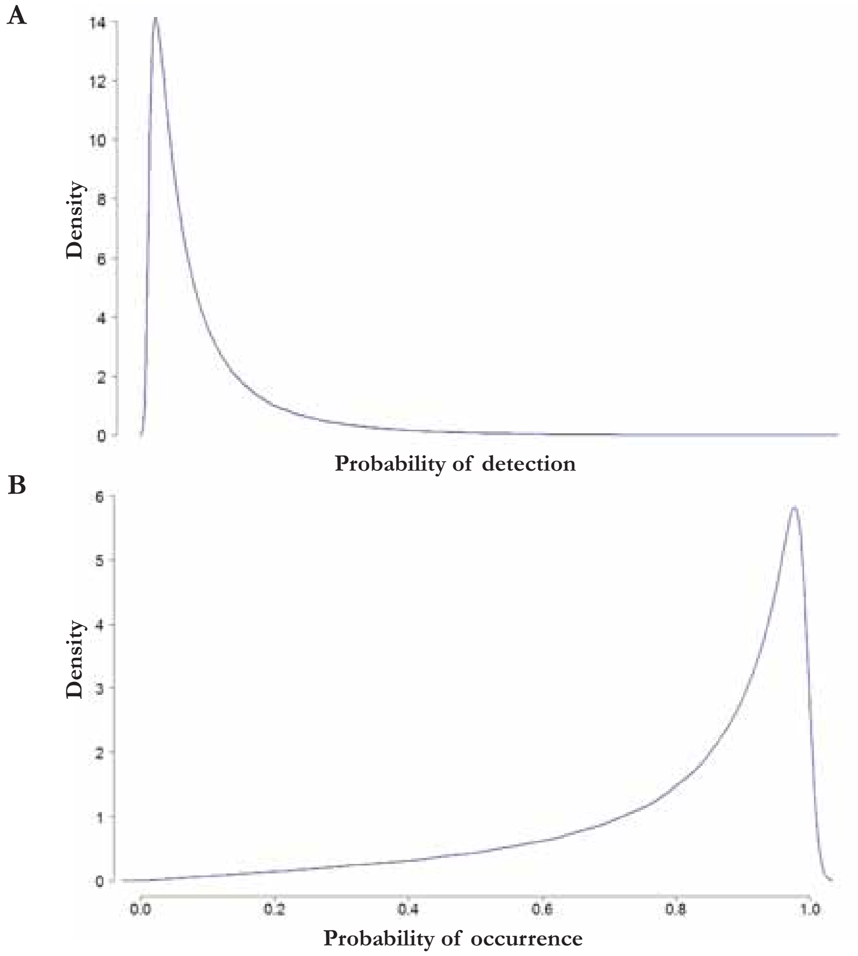

Figure 4.28 Distributions of probabilities of detection (**A**) and occurrence (**B**) of heteropteran bugs in El Maresme shire based on model estimates.

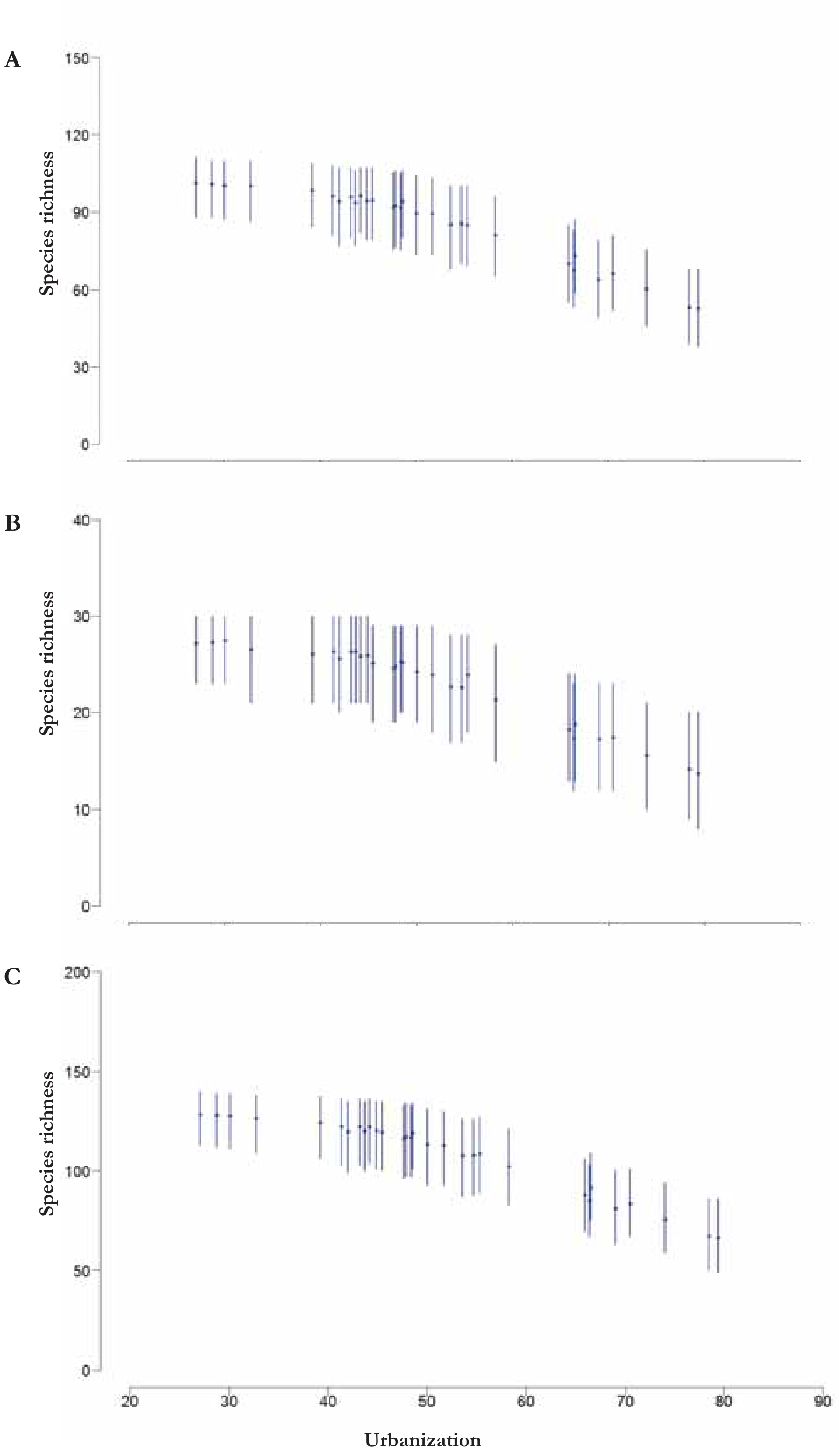

Figure 4.29 Relationships between species richness and urbanization. **A.** Herbivorous guild. **B.** Predatory guild. **C.** Whole community. The solid dots indicates means and the vertical bars 95% credible interval.

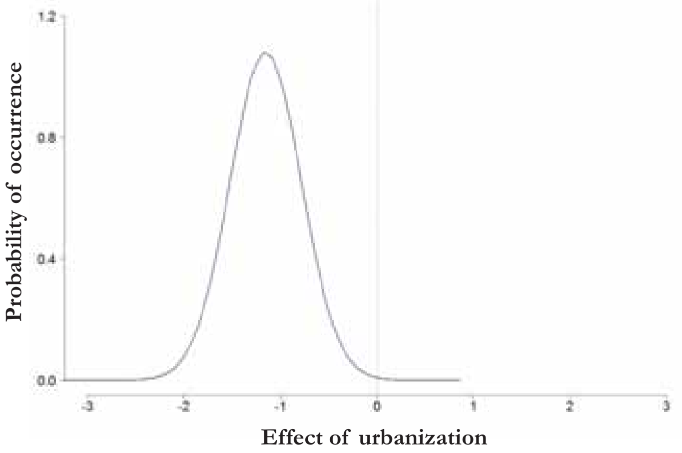

Figure 4.30 Mean community-level effect of urbanization on the occurrence probability of heteropteran bugs (mean=-1.154; CI: -2.013, -0.576).

We used the model's posterior distribution estimates to predict heteropteran bug occurrence probability for 500 values within a reasonable range of the urbanization gradient (20-90). These predictions were then used to illustrate the species-specific relationships between occupancy and urbanization (Figures 31A-E). Urbanization (URB) had a mix of different effects on the species-specific mean probabilities of occurrence. A total of 24 and 7, herbivores and predators, respectively, showed a very large decrease (greater than 8-fold change) in occurrence probability as URB increased over the range of characterized cities (Figure 4.31E). Among the herbivores, 38 species showed a large decrease (greater than 4 to 8-fold change), while 12 predatory species showed the same large response (Figure 4.31D). A total of 25 and 5, herbivores and predators, respectively, showed a moderate decrease (greater than 2 to 4-fold change) in mean estimates of occupancy (Figure 4.31C). The number of herbivorous species showing a slight response (a change in occurrence probability greater than 1.15 but smaller than 2) was 22, while four predatory species showed this same slight response (Figure 4.31B). Finally, three herbivores and two predators showed no response (a change larger than 0.85 but smaller than 1.15) in occurrence probability from the less to the most urbanized city in our study area (Figure 4.31A).

4.6.4 *Estimation of species and family detectability along macroecological gradients*

In total, our models estimated the probabilities of detection of 1,253 heteropteran bug species. The mean, standard deviation and 95% credible interval for these species-specific detection probabilities are given in Table R12 (Supplementary materials), while the same information for the 25 species that showed the highest detection values is given in Table 4.5. In the Iberian Peninsula bioregion, the terrestrial species with the highest probability of detection (mean=0.802; credible interval: 0.693-0.884) was the pentatomid *G. lineatum italicum* (Figure 4.7), while amongst aquatic

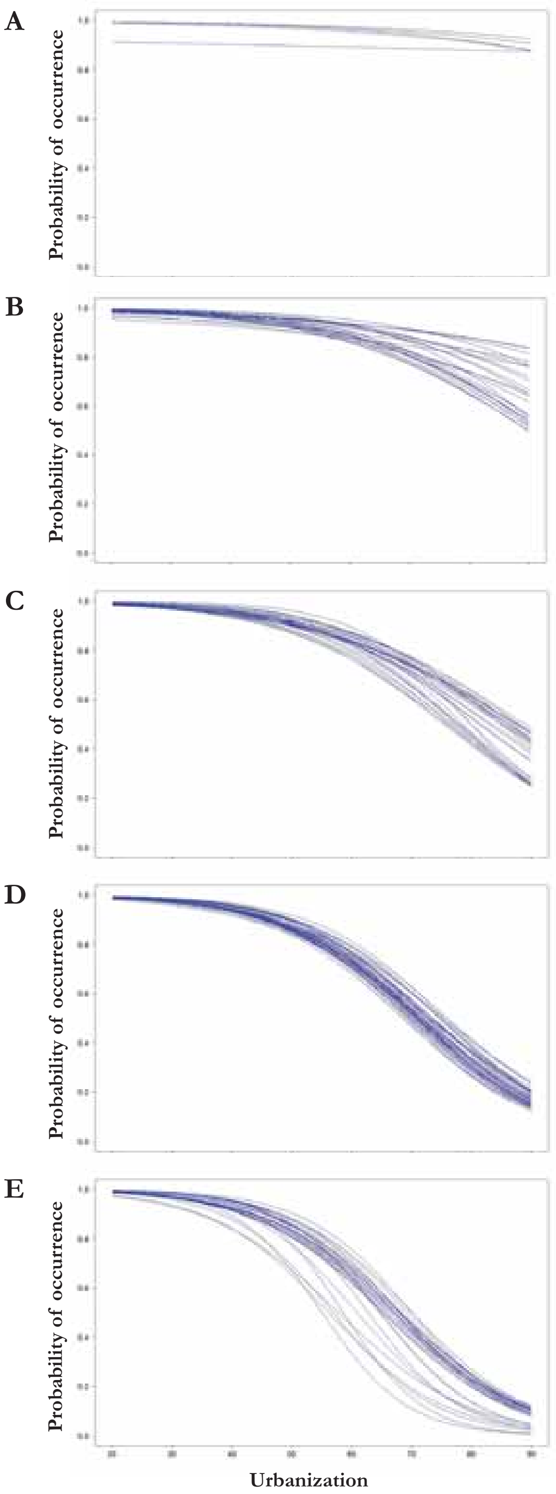

Figure 4.31 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and urbanization. **A.** Species showing no response. **B.** Species showing a slight negative response. **C.** Species showing a moderate negative response. **D.** Species showing a large negative response. **E.** Species showing a very large negative response.

Table 4.5 Mean, standard deviation and 95% credible interval for the species-specific detection probabilities of the 25 species that showed the highest detection values.

Species	Family	Mean	SD	2.5	97.5
Graphosoma lineatum italicum	Pentatomidae	0.802	0.577	0.693	0.884
Carpocoris fuscispinus	Pentatomidae	0.797	0.578	0.683	0.882
Dolycoris baccarum	Pentatomidae	0.771	0.575	0.656	0.861
Spilostethus pandurus	Lygaeidae	0.747	0.571	0.633	0.841
Rhaphigaster nebulosa	Pentatomidae	0.724	0.568	0.610	0.821
Eurydema ornata	Pentatomidae	0.714	0.571	0.594	0.816
Corizus hyoscyami hyoscyami	Rhopalidae	0.690	0.567	0.569	0.792
Camptopus lateralis	Alydidae	0.676	0.569	0.553	0.786
Coreus marginatus marginatus	Coreidae	0.675	0.565	0.557	0.777
Dictyla echii	Tingidae	0.673	0.565	0.557	0.777
Enoplops scapha	Coreidae	0.631	0.563	0.513	0.741
Rhynocoris cuspidatus	Reduviidae	0.628	0.564	0.508	0.740
Leptoglossus occidentalis	Coreidae	0.626	0.563	0.504	0.733
Peirates stridulus	Reduviidae	0.626	0.564	0.506	0.737
Syromastus rhombeus	Coreidae	0.622	0.564	0.500	0.735
Eurydema ventralis	Pentatomidae	0.619	0.564	0.496	0.732
Eurydema oleracea	Pentatomidae	0.613	0.563	0.493	0.723
Spilostethus saxatilis	Lygaeidae	0.611	0.562	0.492	0.720
Rhynocoris erythropus	Reduviidae	0.607	0.562	0.486	0.717
Nezara viridula	Pentatomidae	0.603	0.564	0.481	0.716
Horistus orientalis	Miridae	0.596	0.563	0.476	0.709
Nepa cinerea	Nepidae	0.593	0.565	0.468	0.711
Beosus maritimus	Rhyparochromidae	0.580	0.562	0.462	0.694
Pyrrhocoris apterus	Pyrrhocoridae	0.575	0.563	0.452	0.689
Piezodorus lituratus	Pentatomidae	0.568	0.560	0.450	0.679

species the highest detection probability (0.593; 0.468-0.711) corresponded to the nepid *Nepa cinerea* Linnaeus, 1758 (Figure A3.10).

Through the mean global and mean family-level hyperparameters specified in our models we were able to estimate the whole Iberian Peninsula heteropteran bug fauna probability of detection (mean=0.089; credible interval: 0.069-0.111), as well as separate detection probabilities for the 37 families included in our study (Table 4.6). The three families with the highest probabilities of detection were Coreidae (mean=0.203; credible interval: 0141-0273), Heterogastridae (0.172; 0.096-0.249), and Nepidae (0.161; 0.038-0.426). On the other hand, Saldidae (0.031; 0.017-0.049), Microphysidae (0.031; 0.016-0.055) and Cydnidae (0.049; 0.030-0.073), were the families with the lowest detection values.

Table 4.6 Mean, standard deviation and 95% credible interval for the heteropteran bug fauna-level probability of detection, as well as separate detection probabilities for the 37 families included in the study.

Taxon	Mean	SD	2.5	97.5
Heteroptera	0.089	0.011	0.069	0.111
Family				
Acanthosomatidae	0.071	0.024	0.034	0.127
Alydidae	0.157	0.072	0.050	0.326
Anthocoridae	0.049	0.006	0.037	0.063
Aphelocheiridae	0.067	0.046	0.015	0.187
Aradidae	0.059	0.009	0.041	0.079
Artheneidae	0.110	0.039	0.047	0.200
Berytidae	0.115	0.027	0.068	0.174
Blissidae	0.145	0.042	0.068	0.233
Cimicidae	0.065	0.045	0.015	0.182
Coreidae	0.203	0.034	0.141	0.273
Corixidae	0.065	0.010	0.046	0.086
Cydnidae	0.049	0.011	0.030	0.073
Cymidae	0.116	0.035	0.056	0.192
Dipsocoridae	0.066	0.046	0.015	0.187
Geocoridae	0.095	0.025	0.053	0.150
Gerridae	0.135	0.037	0.070	0.216
Heterogastridae	0.172	0.038	0.096	0.249
Leptopodidae	0.078	0.032	0.032	0.156
Lygaeidae	0.120	0.021	0.083	0.164
Microphysidae	0.031	0.010	0.016	0.055
Miridae	0.050	0.003	0.042	0.058
Nabidae	0.136	0.024	0.092	0.187
Naucoridae	0.114	0.048	0.042	0.228
Nepidae	0.161	0.101	0.038	0.426
Notonectidae	0.134	0.040	0.066	0.222
Oxycarenidae	0.109	0.023	0.069	0.158
Pentatomidae	0.123	0.015	0.094	0.155
Piesmatidae	0.066	0.022	0.032	0.119
Pyrrhocoridae	0.124	0.065	0.036	0.285
Reduviidae	0.083	0.015	0.056	0.115
Rhopalidae	0.149	0.030	0.095	0.215
Rhyparochromidae	0.121	0.009	0.103	0.140
Saldidae	0.031	0.008	0.017	0.049
Scutelleridae	0.073	0.017	0.044	0.109
Stenocephalidae	0.117	0.040	0.052	0.210
Tingidae	0.062	0.008	0.046	0.080
Veliidae	0.078	0.030	0.033	0.151

As shown in Figures 4.32A-C and E and Table 4.7, the mean global effects of area (0.199), altitudinal range (0.600), mean annual temperature (0.204) and population density (0.178) on the detection probabilities of heteropteran bugs were positive, and their posterior credible intervals contained only positive values. On the other hand, the mean global effect of mean annual precipitation (-0.570) was negative, and its posterior interval fell entirely to the negative side of zero (Figure 4.32D and Table 4.7). We estimated the probability of a given macroecological gradient having a positive (or negative) effect on the detection probability of heteropteran bugs as the area under the density curve of its mean global effect hyperparameter to the right (or left) of zero

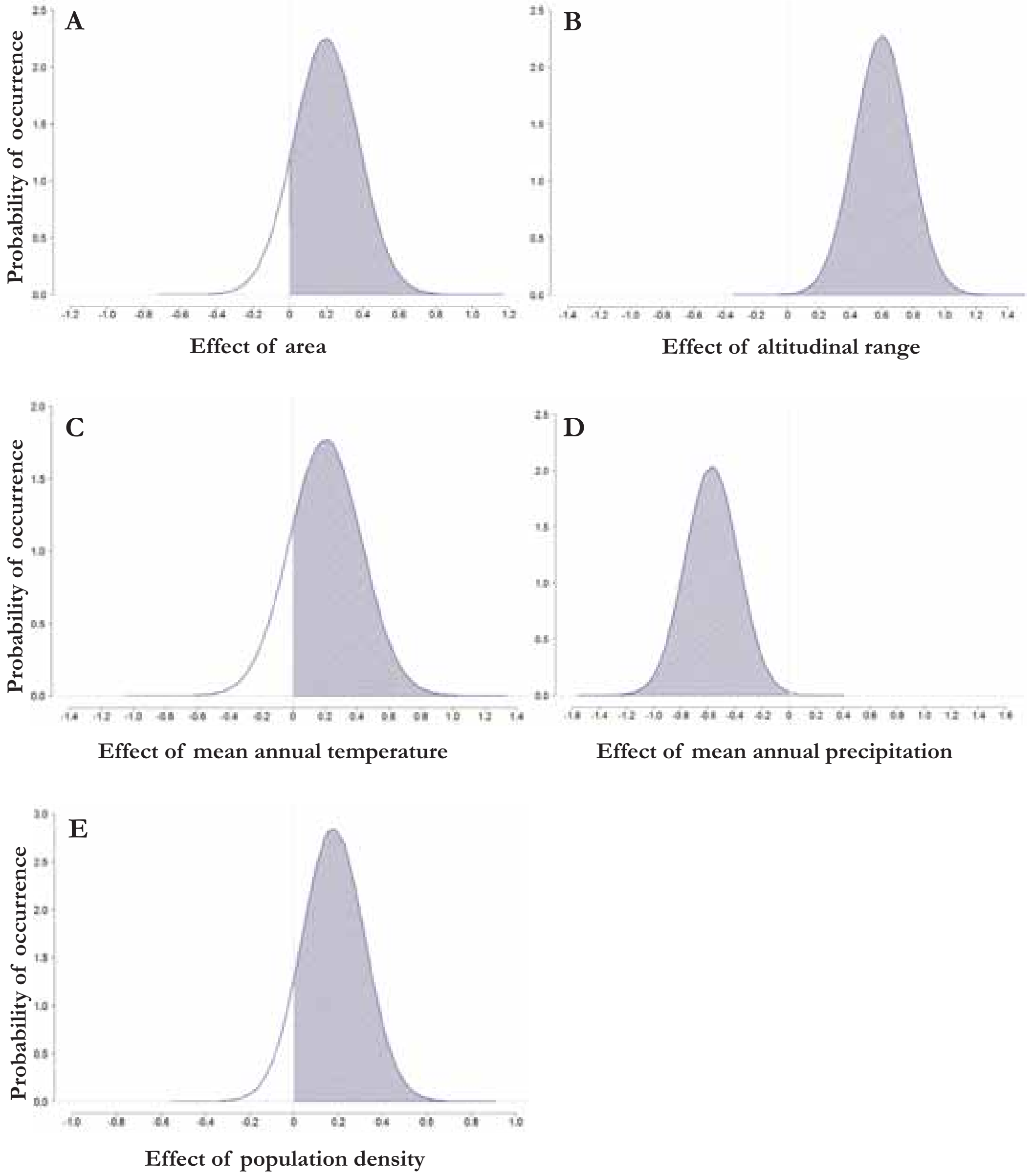

Figure 4.32 Mean community-level effect of area (**A**), altitudinal range (**B**), mean annual temperature (**C**), mean annual precipitation (**D**) and population density (**E**), on the occurrence probability of heteropteran bugs.

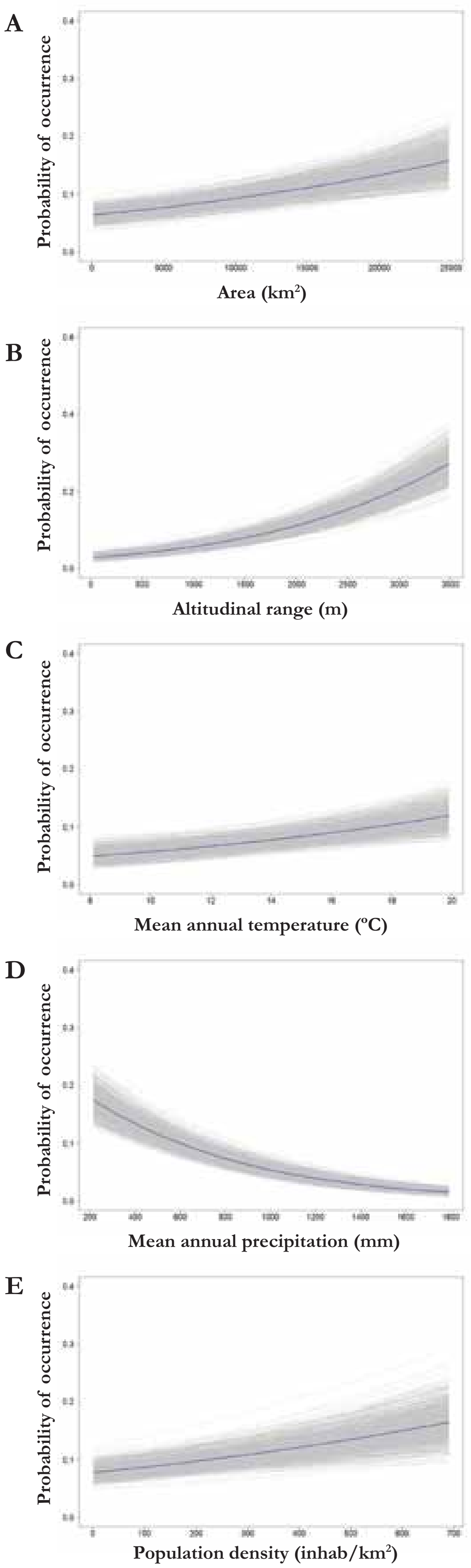

Figure 4.33 Predicted relationships between mean fauna-level occurrence probabilities of heteropteran bugs and area (**A**), altitudinal range (**B**), mean annual temperature (**C**), mean annual precipitation (**D**) and population density (**E**). Blue and shaded lines represent means and uncertainty, respectively.

Table 4.7 Community-level summaries of the hyperparameters for occurrence, detection and the effects of covariates

Community-level hyperparameters	Mean	SD	2.5	97.5
Area				
Mean	0.199	0.041	0.119	0.280
SD	0.178	0.031	0.127	0.248
Altitudinal range				
Mean	0.600	0.042	0.518	0.681
SD	0.176	0.031	0.125	0.247
Mean annual temperature				
Mean	0.204	0.053	0.101	0.312
SD	0.226	0.040	0.159	0.317
Mean annual precipitation				
Mean	-0.570	0.050	-0.669	-0.471
SD	0.196	0.038	0.134	0.284
Population density				
Mean	0.178	0.034	0.111	0.243
SD	0.141	0.024	0.101	0.195

(shaded areas in Figures 4.32A-E). The macroecological gradient with the highest probability of having a positive effect on the detection probability of heteropteran bugs was altitudinal range ($p=0.999$), followed by population density ($p=0.897$), area ($p=0.869$) and mean annual temperature ($p=0.817$). Since we considered families as random-effects, thus specifying into our models family-level effect hyperparameters, we were also able to estimate the effect of the macroecological gradients on each family independently. For each gradient, the means, standard deviations and credible intervals for the 37 family-specific effects on occupancy are presented in Tables R13-17 (Supplementary materials).

We used the models global hyperparameters to predict heteropteran bug detection probabilities for 200 values within a reasonable range of each one of the macroecological gradients. These predictions were then used to illustrate the relationships between the gradients and heteropteran bug detection (Figures 4.33A-E). We illustrated the uncertainty associated with the mean effect by computing and plotting predicted relationships for 1000 random samples taken from the mean effect parameters's Markov chain Monte Carlo draws. This is effectively equivalent to illustrating the whole credible interval.

Likewise, we used the family-level hyperparameters to predict and illustrate the detection probability of each family for 500 values within a reasonable range of the macroecological gradients (Figures 4.34-4.38 and Tables R12-16). All gradients

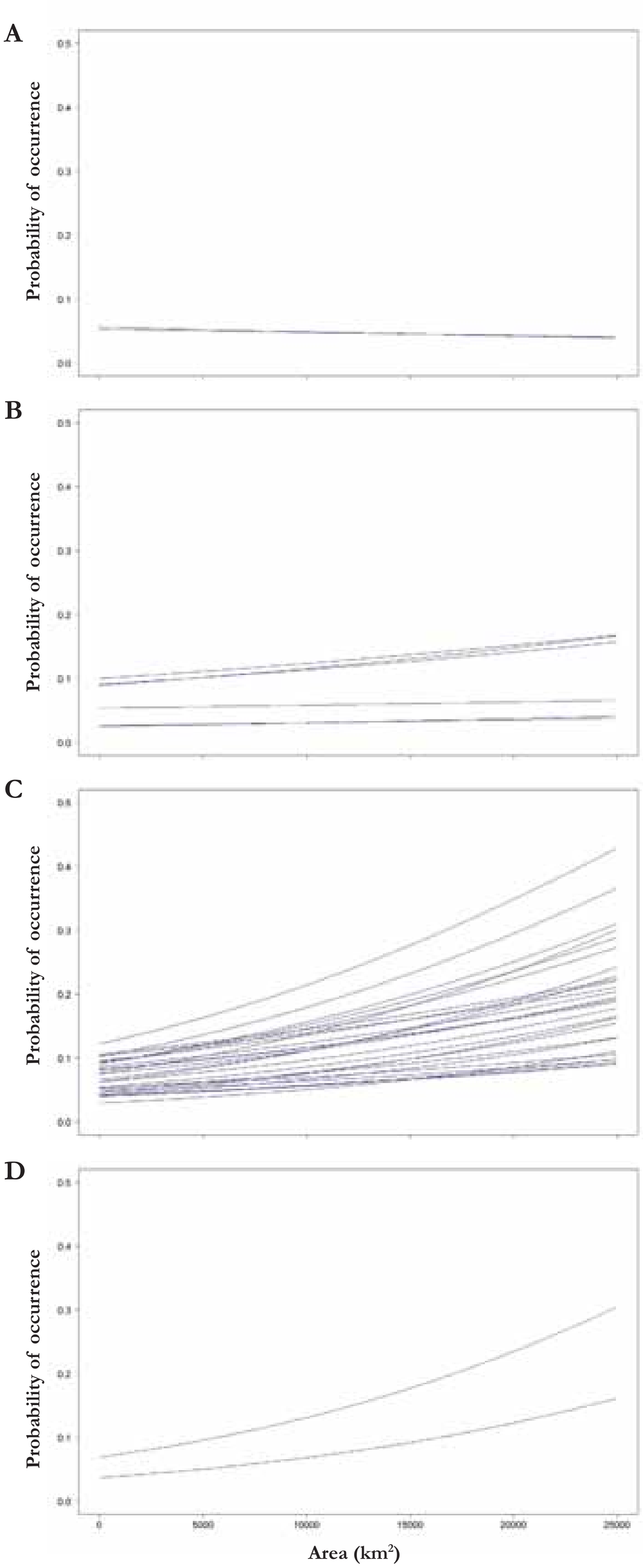

Figure 4.34 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and area. **A.** Species showing no response. **B.** Species showing a slight positive response. **C.** Species showing a moderate positive response. **D.** Species showing a large positive response.

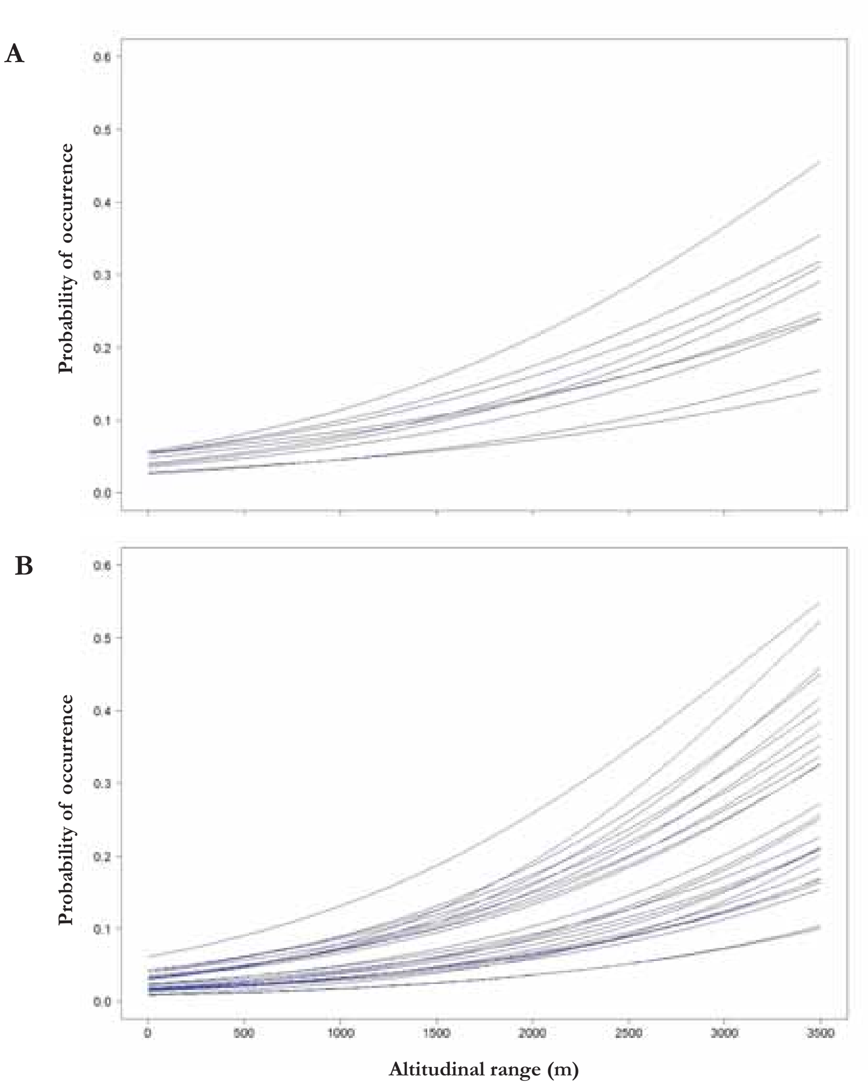

Figure 4.35 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and altitudinal range. **A.** Species showing a large positive response. **B.** Species showing a very large positive response.

showed a mix of different effects on the family-specific mean probabilities of detection. Along the area gradient, two families (Pentatomidae and Corixidae) showed a large increase (greater than 4-fold change) in detection probability as the spatial units increased in size (Figure 4.34D), while 26 and 7 families showed a moderate (> 2 to 4-fold; Figure 4.34C) to slight (> 1.15 to 2-fold; Figure 4.34B) increase, respectively. Two families (Miridae and Anthocoridae) showed almost no response (a change > 0.85 but < 1.15) in detection probability from the smallest to the largest spatial unit (Figure 4.34A). Overall, families along the altitudinal gradient showed distinct positive changes in detection probabilities (Figures 4.35A-B). Of the 37 families, 27 and 10, showed a very large (> 8-fold; Figure 4.35B) to large (> 4 to 8-fold; Figure 4.35A) increase, respectively, in detection probabilities along the

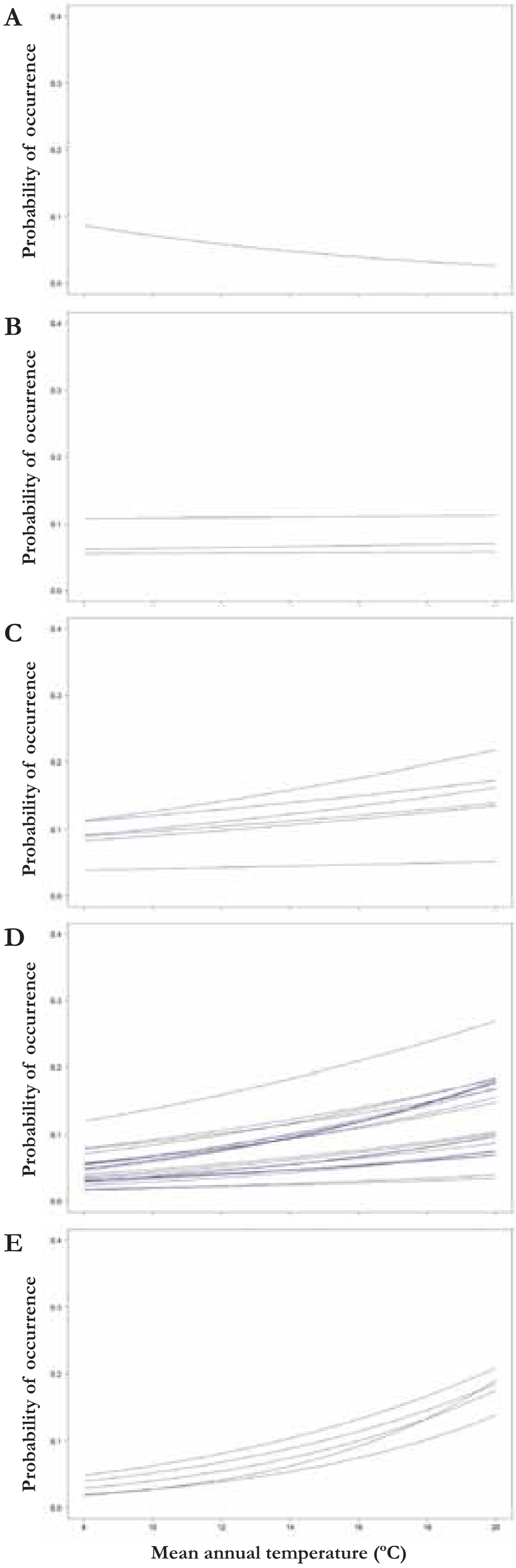

Figure 4.36 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and mean annual temperature. **A.** Species showing a negative response. **B.** Species showing no response. **C.** Species showing a slight positive response. **D.** Species showing a moderate positive response. **E.** Species showing a large positive response.

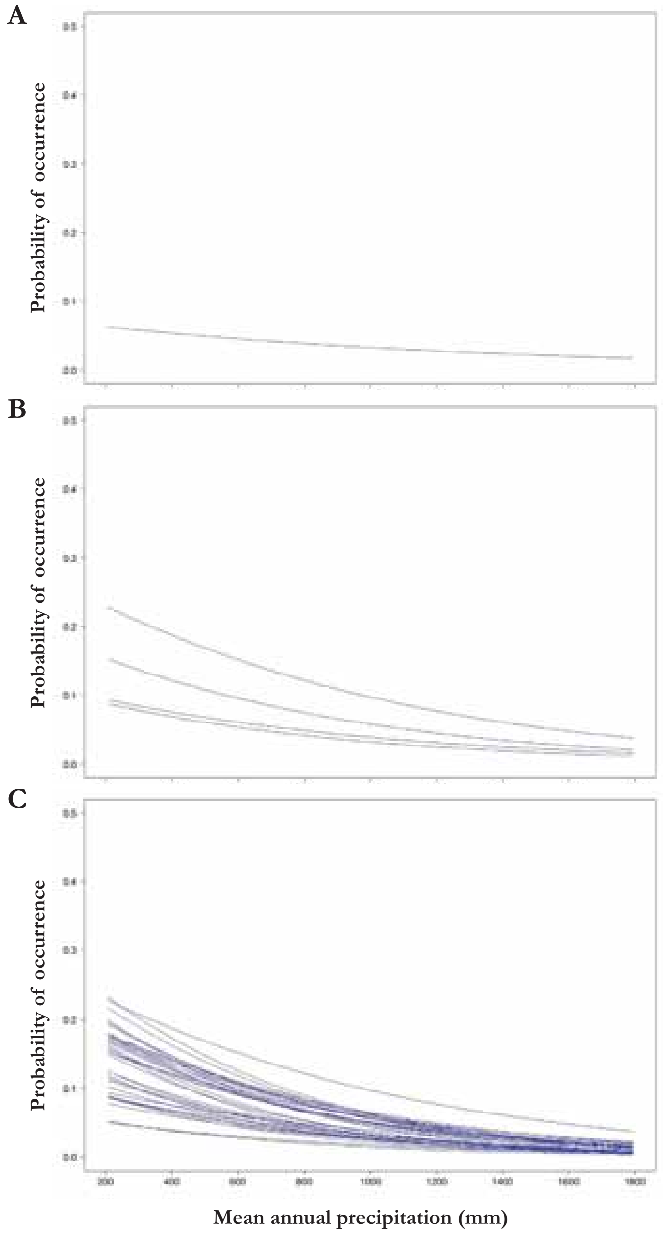

Figure 4.37 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and mean annual precipitation. **A.** Species showing a moderate negative response. **B.** Species showing a large negative response. **C.** Species showing a very large negative response.

gradient. Among these families, the Cydnidae, Rhopalidae and Nabidae showed the strongest responses. The effect of mean annual temperature on the family-specific probabilities of detection varied widely (Figures 4.36A-E). A total of five families (Geocoridae, Lygaeidae, Notonectidae, Reduviidae and Scutelleridae) showed a large increase (> 4-fold change) in detection probability as the spatial units' mean annual temperature raised from its lowest to highest values (Figure 4.36E), while 22 and 6 families showed a moderate (> 2 to 4-fold; Figure 4.36D) to slight (> 1.15 to 2-fold; Figure 4.36C) increase, respectively. Three families (Acanthosomatidae, Aradidae and Cymidae) showed almost no response (a change > 0.85 but < 1.15; Figure 4.36B). Interestingly, as shown in Figure 4.36A, the Miridae showed a moderate negative response (> 2 to 4-fold) as temperature increased along the gradient. All families

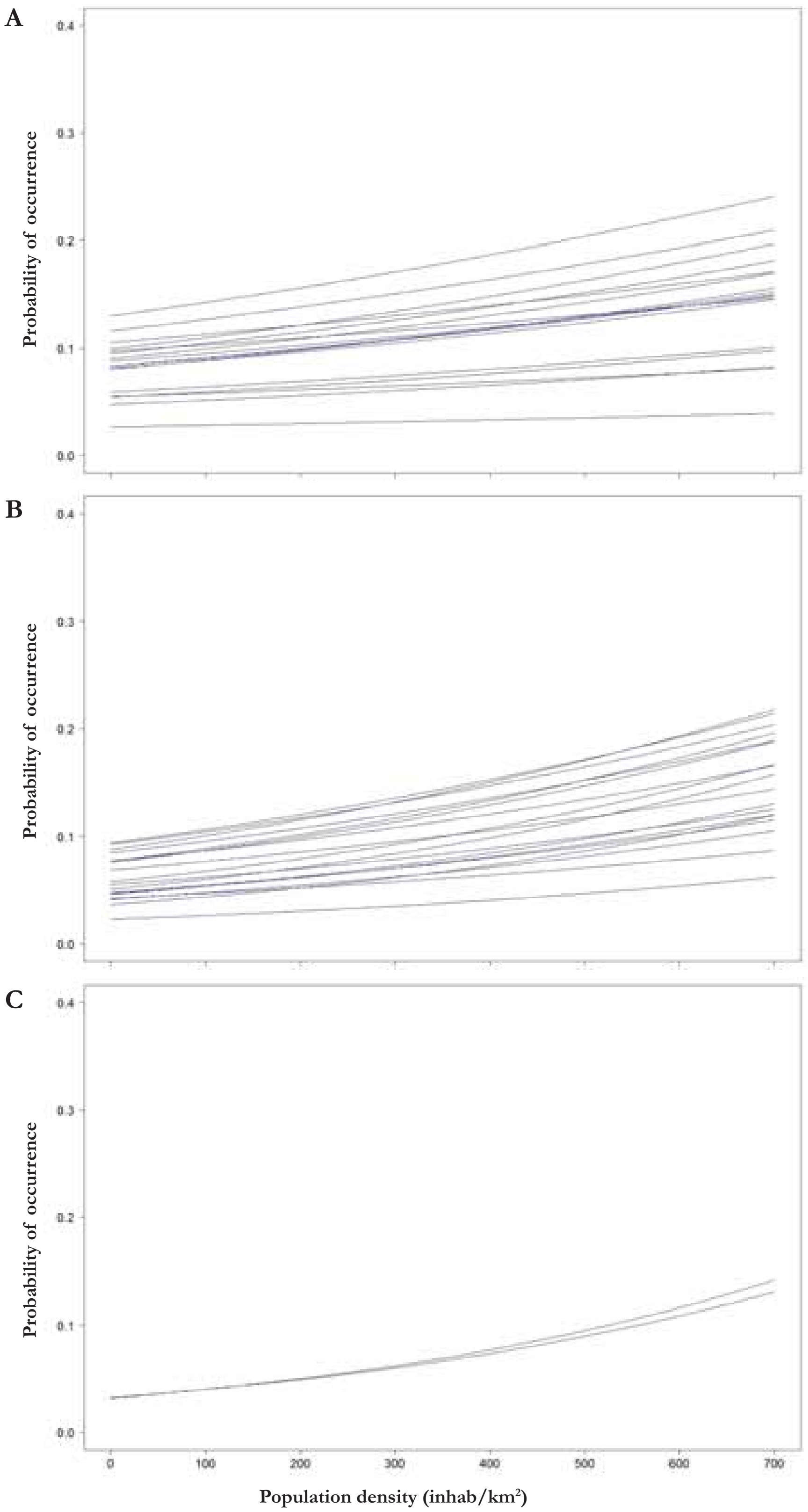

Figure 4.38 Predicted relationships between mean heteropteran bug species-specific probabilities of occurrence and population density. **A.** Species showing a slight positive response. **B.** Species showing a moderate positive response. **C.** Species showing a large positive response.

showed a negative response to the mean annual precipitation gradient (Figures 4.37A-C). A total of 32 families showed a very large decrease (> 4-fold change) in detection probability along the gradient (Figure 4.37C). Among these, the strongest negative responses were shown by the Reduviidae, Pentatomidae and Piesmatidae. Four families (Acanthosomatidae, Coreidae, Cymidae and Aradidae) showed a large decrease (> 4 to 8-fold; Figure 4.37B), while the Miridae showed a moderate response (> 2 to 4-fold; Figure 4.37A). Finally, family-specific mean probabilities of detection showed a mix of positive responses along the population density gradient (Figure 4.38A-C). As seen in Figure 4.38C, two families (Anthocoridae and Cydnidae) showed a large increase (> 4-fold change) in their probabilities of detection from the spatial units with the lowest to highest population densities, while 18 and 17 families showed a moderate (> 2 to 4-fold; Figure 4.38B) to slight (> 1.15 to 2-fold; Figure 4.38A) increase, respectively.

5 Discussion

5.1 On the reciprocal relationships of ecology and taxonomy

5.1.1 *Heteropteran bug hyperdiversity*

Perhaps the most interesting result of this general investigation into the ecology, biodiversity and conservation of heteropteran bugs is that it has gained further evidence of the hyperdiversity of insects. In the present work, we show that the heteropteran bug fauna of Victoria, Australia comprises 438 species. We also provide evidence that the heteropterofauna of the Iberian Peninsula bioregion consists of 1,470 species. These results may be best highlighted by comparing them to the species richness of other better-known emblematic taxa. For example, they show that there are approximately twice as many species of heteropteran bugs in Victoria than amphibian species in the whole of Australia (Chapman 2009) and that there are approximately 50 times as many heteropteran bug species than amphibian species in the Iberian Peninsula (Pleguezuelos et al. 2002). Not surprisingly, our results from smaller areas also reveal high levels of diversity. In El Maresme, a 400 km^2 shire in north-eastern Iberian Peninsula, we documented 323 heteropteran bug species. Also within the Iberian Peninsula, we found 59 species present in the 17 km^2 municipality of Cillaperlata (Burgos) (Figure 5.1). Likewise, 31 species were observed in a narrow

Figure 5.1 Fields and forested-hills characterize the landscapes surrounding the town of Cillaperlata (Burgos, Castille-Leon, Spain). Source: original.

Figure 5.2 The Jardinet de l'Om (Elm Garden) is a student-managed urban garden and vegetable plot located in the densely-urbanized neighbourhood of Les Corts (Barcelona, Catalonia, Spain). Source: Rafael Arocha

herbaceous margin in the village of Artekona (Gordexola, Vizcaya) and in the small 'Jardinet de l'Om' urban garden (University of Barcelona, Faculty of Biology, Les Corts, Barcelona) (Figure 5.2). These results strikingly highlight that within the Iberian Peninsula the species richness of heteropteran bugs in small herbaceous or garden plots may be equally as high as that of amphibians in the whole 582,000 km^2 bioregion.

During the present thesis, we collected and worked with approximately 17,000 heteropteran bug specimens, which translated into over 4,000 new field records of 512 different species. A total 418 of these species were collected in 334 localities within the Iberian Peninsula. Reviewing the recent patterns of how the rich diversity of heteropteran bug species has been documented in the Iberian Peninsula revealed many interesting and important predoctoral investigations that are worth mentioning. Vázquez (1985) worked with approximately 4,000 collection and new field specimens to review the 62 species of Alydidae, Coreidae and Rhopalidae present in the bioregion. Likewise, working with the Miridae, Goula (1986) collected over 5,000 new field specimens belonging to 119 different species to review this family in Catalonia. Regarding aquatic taxa, Baena (1980), working in the 'Sierra de los Santos y Sierra de Córdoba' (Córdoba), collected approximately 2,000 specimens and found them to belong to 34 gerromorphan and nepomorphan species. Similarly, working in the province of Madrid, López (1998) collected approximately 6,500 specimens of aquatic gerromorphan and nepomorphan taxa belonging to 45 species. More recently, Costas (2004) conducted a study of the family Lygaeidae (that at the time included also families Cymidae, Heterogastridae, Geocoridae, Oxycarenidae and Rhyparochromidae) in the 'Sierra de Gredos' (Ávila), and found 69 species among the approximately 3,800 collected specimens. Gessé (2004) studied the heteropterofauna of the Garraf Natural Park (Barcelona), collecting over 3,000 specimens and documenting 77 species. Last but not least, E. Ribes (2004) conducted a study of the heteropteran bug fauna of the Collserola Natural Park (Barcelona) revealing 144 species among approximately 5,000 collected specimens. Knowledge of the rich diversity of the Iberian Peninsula heteropterofauna has undoubtedly also benefited by the faunistic contributions throughout the years of a large list of dedicated taxonomists, including the more recent works of the researchers just mentioned. Unfortunately, the proper acknowledgment of the large body of work generated by them overreaches the conceptual context of the present discussion. However, no discussion of the recent advancements in the understanding of Iberian Peninsula heteropteran bug biodiversity would be complete without a recognition of the abundant and diverse contributions of J. Ribes (reviewed in Goula 2011). Furthermore, I believe it is also worth mentioning the pivotal contributions of S. Pagola-Carte in the Basque Country (Pagola-Carte et al. 2003, 2005), J. Pérez and F. Prieto in Galicia (Pérez and Prieto 2009), J.M Grosso-Silva in Portugal (Grosso-Silva and Soares-Vieira 2009, Grosso-Silva 2004) and N. Nieser and C. Montes for documenting the aquatic heteropteran bugs in the whole Iberian Peninsula bioregion (Nieser and Montes 1984).

Our results also show that the 95 species we found in Victoria account for over 20% of the region's known heteropterofauna. One interesting aspect of our results is that they indicate that in Victoria there are twice as many species of Pentatomidae than of Miridae. This finding is consistent with the Australian regional estimates of species richness documented in Cassis and Gross (1995, 2002) and provided by Henry (2009). Importantly and interestingly, this pattern contradicts the patterns observed for the Nearctic, Palearctic and the World, which show 8.7, 3.3 and 2.1 times more species, respectively, of Miridae than Pentatomidae. Our own estimates indicate that in the Iberian Peninsula there are over five times as many species of Miridae than Pentatomidae. However, given the strong taxonomic impediment the Miridae have historically experienced in Australia and other regions of the Southern Hemisphere (Cassis and Schuh 2012), this Australian departure from the expected Miridae pattern of species richness, including the one observed in our study, should come as no surprise. Evidence for this impediment has been robustly put forward by many findings from the 'Plant Bug Inventory project' (Cassis 2008, Tatarnic & Cassis 2008, Schuh and Pedraza 2010, Menard and Schuh 2011). In fact just two of the revisions mentioned (Cassis 2008, and Schuh and Pedraza 2010) contributed to increase the number of know Australian Miridae by 30%. I believe the 16 Miridae species collected in our study that await proper species identification or description are further proof of this documented taxonomic impediment, and that, in combination with present and future research, they will contribute to demonstrate that the present ratio of Pentatomidae-Miridae in Australia may be an artifact resulting from inadequate sampling.

5.1.2 Opening taxonomical doors with the right keys

An essential assemblage of tools that allowed us to properly identify the 512 heteropteran bugs reported in this thesis were diagnostic dichotomous keys. Fortunately, the Palaearctic, Euromediterranean and Iberian entomological literature provide enough diagnostic keys (see Table M2 in the Supplementary materials) to confidently identify almost every species present in the Iberian Peninsula bioregion. Although Australia is associated with the strong taxonomic impediment just discussed (Cassis and Schuh 2012) and, consequently, diagnostic keys for some taxa may be lacking, the entomological literature is still rich in works that allow for the identification of heteropteran bug species and higher taxa present in Victoria (for examples see Gross 1975, Carver et al. 1991, Malipatil 1994, Brailovsky 2007). However, through the use of these keys to identify species both in the Iberian Peninsula and Australia, we identified several important issues (eg, inclusion of non-relevant taxa, non-inclusion of recently described taxa and/or synonymies, low degree of observability, awareness of in-situ photographic records, exclusive use of local language) that may require special attention. In an attempt to explore and address some of these issues, we developed during the present thesis over 150 dichotomous keys. As has been previously stated, a broad goal of this work is to couple faunistic and ecological research to the benefit of conservation. Hence, the presentation of all our developed keys, a task more associated with a pure taxonomical effort, fell out of the contextual framework of our study.

Nevertheless, I judged it pivotal to present at least some examples of our keys (Appendices II to VI), and to discuss the potential benefits of these keys to contribute to heteropteran bug identification to the species or higher taxa level. First, keys including only taxa pertinent to the region under study may become not only shorter but, importantly, less complex. For example, Our *Key to the tribes of Miridae from the Iberian Peninsula* includes only 13 of the total 35 currently recognized tribes for the world (Cassis and Schuh 2012) and our *Key to the families of Heteroptera from Victoria* includes only 48 out of the 68 families present in Australia (as documented in Henry 2009). Second, keys become current and accurate when they incorporate the most recent taxonomical information. Two examples regarding our new keys come to mind: (1) the *Key to the genera of Rhyparochrominae from the Iberian Peninsula* allows the identification of *Tempyra biguttula*, a species belonging to a genus and tribe previously unknown to both the Iberian Peninsula and the Palearctic (Baena and Torres 2012, Aukema et al. 2013), and (2) the *Key to the species of Deraeocoris from the Iberian Peninsula* allows for the identification of *Deraecoris flavilinea*, a recent addition to the known mirids from the Iberian Peninsula (Gessé 2011, Goula and Mata 2011, Vivas 2012). Third, keys focusing on character observability instead of phylogentic coherence may become more practical for identification purposes. For example, the traditional approach to identification of rhyparochromid genera involves identifying specimens first to tribe level, a step that relies strongly on the use of characters that suffer from a low degree of observability. Our *Key to the genera of Rhyparochrominae from the Iberian Peninsula*, by proceeding directly to the genus level, specifically avoids the use of these poorly-observable characters (eg, trichobothria and spiracles), focusing rather on other well-observable ones (eg, pronotum and scutellum). Fourth, keys become more flexible when they include characters that are equally efficient in both 'under the microscope' and 'photographic or in-situ' identification. This becomes specially relevant if we consider that by identifying a species directly in the field or by photographic methods (instead of by means of a physical specimen) we are effectively contributing to address important conservation issues (eg, over-collection, collection of rare and/or endangered species). For example, by relying extensively on characters observable on the dorsal side of the body, our *Key to the species of Deraeocoris from the Iberian Peninsula* allows for the photographic identification of almost all *Deraeocoris* species present in the bioregion (the exceptions are *D. ribauti* and *D. morio* that require the observation of the small ventrally-located openings of the metathoracic scent-glands). Last, keys written in English, or at least offering an English translation of their couplets, may become more accessible to the wider scientific community. Here, we followed this approach and specifically wrote all our keys in the English language. In this respect, we applaud the recent efforts of the editors of the 'Faune de France' series to include English translations of their otherwise French-based keys (for examples see Ribes & Pagola-Carte 2013).

5.1.3 *Is a photo worth more than a 1,000 bugs?*

One of the most specific aims of the present thesis was, perhaps, that it explicitly incorporated the use of in-situ photography and photographic biodiversity web resources. Our results clearly indicate that they notably meet the challenge of expediting conservation-oriented research and engaging the general public in the conservation of nature. Both the *Catalog of the Heteroptera from the Iberian Peninsula* and *Estimation of species and family detectability along macroecological gradients* studies benefited from over 23,000 photographic records contributed to biodiversity web resources by over 750 photographers. These are no trivial numbers, especially when compared to the number of available bibliographical records (approximately 14,000) and the number of contributing papers (approximately 440) we documented for these same studies. Given the ease and cost effectiveness by which new photographic material is collected in the field, and, more importantly, given the eagerness and motivation of amateur and professional photographers alike to engage in nature-related endeavors, I anticipate that the scientific importance of in-situ photography as channeled through web biodiversity resources will continue to increase.

However, in order to effectively contribute to scientific research some critical issues regarding in-situ photographs and web resources must be addressed. First, as noted by Goula et al. (2012), an insect 'photographic record' must hold enough metadata (ie, date, location, habitat) to make it comparable to a classic 'field record'. In this respect, it is worth mentioning the working methodology of the web citizen's biodiversity platform Biodiversidad Virtual (2013), which requires their members to fill in a series of metadata fields before their images can be uploaded. Other web-based projects should implement similar procedures to guarantee that their photographic material may be effectively used as a photographic record. For example, Flickr photos to be picked-up by the automated algorithm used by the Encyclopedia of Life (2013) could be required to also hold a machine-tag for 'date' and 'location'. Secondly, given that certain characters or character states are very difficult (perhaps impossible) to observe, identification to species level by means of in-situ photographs is frequently neither possible nor desirable. Taxonomist used to physical specimens 'under the microscope' are often faced with this same dilemma when essential characters for species identification are not possible to observe (eg, because of missing appendages, body parts or genital segments). For example, taxonomical revisions conducted by Ribes and Pagola-Carte (2009, 2013) suggest that species belonging to the *Carpocoris* genus (Figure 3.4) present in the Iberian Peninsula might only be correctly separated from each other by noting the state of a very small morphological character present in a genital structure, hence not observable in an in-situ photograph. Unfortunately, 60% of the photographic records attributable to the *Carpocoris* genus contributed by 'Biodiversidad Virtual' were identified to species level (the other 40% were more suitably labeled as either *Carpocoris* sp. or *Carpocoris cf. fuscispinus*). Incidentally, *C. fuscispinus* turned out to be the top recorded species by the platform. I therefore suggest that special attention should be paid by web-based taxonomists to the limitations inherent to in-situ photographs when identifying portrayed taxa to species level. Finally, in order for in-situ photography to fully contribute to an accessible and transparent modernization of taxonomy, ownership of photographic material used to generate photographic records should be released to the commons. A good example is the 'Encyclopedia of Life', which only stores and shares media from creators that have previously given their work a 'Creative Commons' license.

As long as the issues previously discussed are taken into account by photographers and biodiversity web resources alike, in-situ photographs may substantially expedite conservation-oriented research, while simultaneously engaging the general public in activities relevant for the conservation of nature. In the words of Marshall (2008), "Digital insect collecting—and a contribution to the democratization of insect taxonomy—is truly within everyone's reach". Of course, insect photographs may be quite aesthetically pleasing by themselves, and may be ideally suited to visually communicate concepts and ideas for which words might fall short or might be lacking, as I can only hope is reflected in the many heteropteran bug images that illustrate this work.

5.1.4 Cataloging biodiversity

Faunistic syntheses such as catalogs and datasets are essential for documenting where species occur and how they are distributed. For each taxa they document, they may also provide valuable information on taxonomical nomenclature, authorship, synonymies and older name combinations. Importantly, they should provide data regarding the bibliographic, photographic, and new field records that have led to the documentation of the distributional patterns of the species under consideration. Moreover, faunistic syntheses may be central to identify potential conservation issues regarding the species present in a given region. It is for these reasons that we explicitly coupled the ecological investigation conducted in the present thesis with the development of faunistic catalogs and datasets for the regions that were studied.

As mentioned above in *Heteropteran bug hyperdiversity*, our *Heteroptera from Victoria* dataset shows that the heteropteran bug fauna of Victoria, Australia comprises 438 species. This indicates that since the publication of the "Zoological Catalogue of Australia" (Cassis and Gross 1995, 2002),which documented the occurrence of 417 species in Victoria, 21 species have been added to the region's heteropterofauna. Among these added species were *Crompus oculatus*, *Koscocrompus obscurus*, and *Melanacanthus scutellaris* that we reported here for the first time in the Victoria region. This large increase suggests that our cataloging of the Victoria heteropterofauna, and by extension of the Australia heteropteran bug diversity, is by no means a completed task. In fact, not included in our *Heteroptera from Victoria* dataset were 57 taxa that are pending proper species identification or description. This highlights that more research efforts may be needed to achieve an accurate knowledge of the occurrence and distributional patterns of heteropteran bugs in Victoria.

Results from our faunistic syntheses also highlight the importance of combining different sources of information for obtaining a more focused understanding of the ranges throughout which species are distributed in a given region. For example, in

Pyrrhocoridae from the Iberian Peninsula we show that knowledge regarding the spatial distribution of *Pyrrhocoris apterus* (Figure 3.7 and A3.7C) in the Iberian Peninsula bioregion may be strongly increased (by over a 100%) when photographic records, new field records and observations are added to pre-existing distributional knowledge as documented exclusively by bibliographical data. I envision that this approach may be strengthened as more new biodiversity data becomes available from global biodiversity open-access networks such as the Global Biodiversity Information Facility (2013), regional biodiversity data projects such as the Biodiversity data bank of Catalonia (2013) and 'Citizen's Science' projects such as the 'Bugs Count Survey' coordinated by the Open Air Laboratories (2013).

Importantly, the catalogs and datasets presented in this work resulted from the attempt to bridge the gap between taxonomy and conservation by explicitly linking faunistic survey efforts with the monitoring of state variables relevant to ecological quantitative research. For example, in *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* our interest in quantifying the effect of unmanaged herbaceous and of complexly-structured vegetation within golf courses on insect biodiversity led us to monitor heteropteran bug species richness and species-specific patterns of occupancy. This in turn led us to the detection of 76 heteropteran bug species, including *C. oculatus*, *K. obscurus* and *M. scutellaris*, which are known in Victoria exclusively by the observations made in some of the south-east Melbourne golf courses that were part of the study. Thus, these species were discovered, identified and catalogued precisely thanks to quantitative research efforts aimed at their conservation. Another interesting example is provided by the case study *Effects of urbanization on occupancy and species richness*. In this case, our concern with exploring how the urbanization pattern of a whole shire may be influencing insect biodiversity drove us to monitor municipality-level patterns of species richness as well as the occurrence of individual species. As a consequence, we were able to record occurrence data for over 44% of the known heteropteran bug species of 'El Maresme' shire in north-east Iberian Peninsula. Although all but one species (the blissid *Ischnodemus sabuleti*) were already known for the region, this study provided the parallel faunistic study *Heteroptera from El Maresme* with over 50% of its municipality-level distributional records. Our results indicate that these two coupled studies complement each other considerably and share key findings. The quantitative research, for example, predicts that Mataró, the municipality with the highest degree of urbanization, should have the lowest value of heteropteran bug species richness, as well as the lowest probabilities of occurrence for most species. On the other hand, the faunistic catalog documents this same municipality of Mataró to present the second largest number of species of the whole shire. However, a closer inspection of the faunistic data shows that most of the species that contribute to the high diversity of Mataró have not been re-recorded in the municipality in over 100 years. Hence, while the ecological research quantifies the effect of a landscape-level perturbation that may drive the extinction of species, the faunistic research is concomitantly indicating that these extinctions have in fact taken place and is able to highlight exactly which species and higher taxa have been affected.

Arguably, one of the most original contributions of the present thesis is our calculation that the Iberian Peninsula bioregion heteropterofauna consists of 1,470 species. This result was made possible by our efforts to use taxonomy as a coupling force between faunistics and quantitative ecology to the benefit of pure and conservation science. In fact, the faunistic synthesis *Catalog of the Heteroptera from the Iberian Peninsula* was specifically designed to provide detection/non-detection data for each species at the district/province level to the quantitative case study

Estimation of species and family detectability along macroecological gradients. Results from both these studies suggest that faunistic syntheses extending over whole bioregions may effectively contribute to overcome the 'Wallace shortfall' associated with large scale distributional data, which is exactly the type of data most needed by macroecology and conservation biogeography studies.

In its current state of development, the *Catalog of the Heteroptera from the Iberian Peninsula* is by no means a work near completion. Although our results indicate that over 14,000 bibliographical records were available to develop the catalog, I believe a deeper exploration of the literature may reveal even more records. Likewise, it is reasonable to expect that new photographic records will be added frequently to the dataset. At a slower pace, new field surveys, conceived to address specific Wallace shortfalls within the Iberian Peninsula, may contribute to overcome the faunistic impediment associated with some districts/provinces within the bioregion. Possibly, the single most effective tool that could guarantee a constant flow of data to the catalog would be the establishment of a research-oriented 'Heteropteran Bug Monitoring Scheme', analogous to existing insect monitoring programs such as the Catalan Butterfly Monitoring Scheme (2013) and the butterfly component of Biodiversity Monitoring Switzerland (2013). Finally, there is the issue of scale. Intrinsic to the development of a faunistic catalog is the challenge of choosing an appropriate spatial resolution for the distributional data. Large ecozone-scaled catalogs, for example, the "Catalog of the Heteroptera of the Palaearctic Region" (Aukema and Rieger 1995, 1996, 1999, 2001, 2006, Aukema et al. 2013), may have their spatial resolution fix at a certain level (eg, country level). Other works, for example, the "Catàleg dels heteròpters de Catalunya" [Catalog of heteropteran bugs from Catalonia](Ribes et al. 2004) are set to a mix of variable resolutions (ie, 10 x 10 UTM quadrats, provinces, shires, mountain ranges). As our methodology shows, our *Catalog of the Heteroptera from the Iberian Peninsula* fixed its spatial resolution at the district (Portugal) and province (Andorra and Spain) levels. Hence, by moving from the country to the district/province level, our 'Iberian Peninsula' catalog effectively increases the resolution of the 'Palearctic' catalog. As these finer resolutions are needed to investigate the distributional patterns of rarer species occupying narrower ranges, I will argue that further work on the *Catalog of the Heteroptera from the Iberian Peninsula* should focus on reducing its spatial resolution to the shire and municipality levels.

Through their contribution to the documentation of the occurrence and distribution of species at various spatial scales and to the identification of potential treats to biological diversity, our catalogs and dataset highlight the pivotal role that taxonomy and faunistics play in the conservation arena. For this role to be played out effectively, however, the role of quantitative ecology should also be recognized. Bridging the gap between taxonomy and conservation might require faunistic survey efforts to be linked to the monitoring of state variables relevant to ecological research.

5.1.5 *Hierarchical models, Bayesian inference, detectability and the challenge of scale*

In his 1928 paper to the journal ‘Ecology’ titled “The quantitative analysis of environmental factors” R. Chapman wrote:

> *Throughout the history of various branches of science there can be seen a trend from the field of relatively inexact pure description to the field of relative exactness involving quantitative methods and mathematical calculations.*

In the present work, I have followed this trend into *the field of relative exactness* guided by the hierarchical view (Royle and Dorazio 2008), an approach to quantitative ecology that focuses on simultaneously accounting for the stochasticity associated with the ecological process driving the observations of the living systems under study and the stochasticity associated with the observation process that gathers the data. Specifically, I have explored and illustrated here the use of a hierarchical community model: the multi-species site occupancy model (msSOM). Results from the four case studies presented in this work suggest that the msSOM is an effective quantitative tools for the estimation of species-specific probabilities of detection and occurrence, from which the size of the community (ie, species richness) may also be estimated. Because these parameters were considered random effects, it became possible to describe their distribution using hyperparameters. Hence, results from the case studies also include the estimation of community-level probabilities of detection and occupancy. The case study *Estimation of species and family detectability along macroecological gradients* was exceptional in this respect as it incorporated hyperparameters to describe the distribution of species-specific random effects at both the family and whole heteropterofauna levels. This highlights the flexibility of hierarchical models for the estimation of parameters at multiple levels (Gelman and Hill 2007). Another powerful characteristic of the msSOM is that it utilizes the collective community data to provide information on the occurrence probability for all observed species, resulting in improved analyses of the community and increased precision in species-specific estimates of occurrence (Dorazio and Royle 2005, Dorazio et al. 2006, Kéry and Royle 2008, Zipkin et al. 2009, 2010). Considering that the heteropteran bug communities studied here generally included many rare species, this characteristic of the msSOM was very useful to our analyses. Finally, another advantage of the msSOM, and of any hierarchical linear model in general, is the ease by which covariates may be specified in the linear predictors of both the ecological and observation process levels. For example, in *The effect of landscape functional heterogeneity on vineyard biodiversity* we assessed the effect of sampling methodology on species-specific detection probabilities, and were able to determine that in vineyards the flight intercept trap was consistently more efficient than the pitfall trap for surveying heteropteran bugs. For these reasons, I believe that hierarchical models are flexible and powerful tools for conducting quantitative ecological research and that the multi-species site occupancy model has a substantial potential to advance our knowledge of insect species and communities.

An important quantitative goal of the present thesis was to demonstrate the use of the Bayesian mode of inference. Therefore, Bayesian methods were exclusively used to make inferences on the parameters specified into the multi-species site-occupancy models. As demonstrated by the models presented in this work, Bayesian methods consider all unknown parameters as random variables described by probability distributions. Estimation of a given parameter results in a posterior distribution that provides the mean and, most importantly, the uncertainty associated with the estimate. This exceptional characteristic of the Bayesian mode of inference allowed us to focus the quantitative research presented here on estimating effects sizes and providing a measure of the precision of those estimates. In *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*, for example, we report and illustrate the effect of vegetation density on the mean community-level probability of occurrence of heteropteran bugs, and, by means of this hyperparameter's 'area under the curve', we were also able to report the actual probability of the covariate having a positive effect on heteropteran bug occupancy. Likewise, in *Estimation of species and family detectability along macroecological gradients*, we used our Bayesian-estimated posterior distributions, to report and illustrate the effect of macroecological gradients on the mean detection probability of the whole Iberian Peninsula heteropterofauna and to compute the actual probabilities of the covariates having a positive or negative effect on detection. In this latter example, however, we took further advantage of the Bayesian versatility to, by means of random samples taken from the parameters' Markov chain Monte Carlo draws, represent the whole credible interval associated with the predicted relationships between the covariates and detection. To summarize, the Bayesian mode of inference provides us with a powerful tool to conduct quantitative research and to communicate these results in an effective and clear manner to those responsible for management and policy decisions regarding the conservation of biological diversity (Wade 2000, Ellison 2004, Clark 2005, McCarthy 2007, Royle and Dorazio 2008, Kéry 2010). I believe that the preceding examples, and, in general, the results presented in this work, demonstrate some benefits of Bayesian inference for addressing ecological, biodiversity and conservation questions. Nevertheless, I wish to remark that other potential benefits and challenges of the Bayesian approach were not explored here. For example, I did not combine the empirical data with prior knowledge about the systems under study. In this respect, I assumed a complete ignorance of the system, and therefore specified this ignorance using flat non-informative priors. I anticipate that future conservation-oriented research focusing on insect communities could greatly benefit from exploring ways to incorporate into their quantitative models prior knowledge about the systems they wish to investigate.

Given the importance of accounting for the uncertainty inherent to the methods that we use to observe and measure the living world, it is remarkable how often studies overlook spatial variation and detectability as sources of error (Boulinier et al. 1998, Yoccoz et al. 2001, McKenzie et al. 2002, Kéry and Schmidt 2008, Kéry 2011, Wintle et al. 2012). One of the specific quantitative goals of the present work was to explicitly account for the stochasticity associated with spatial variation and the imperfect detection of species. As evidenced in the case studies, we accounted for spatial variation by including some degree of randomness in the selection process of the sampling plots. For example, in *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* case study, the 'woodland' plots were selected in a GIS environment using a random point generator. Likewise, the survey protocols used in this thesis were specifically designed to include spatial and/or temporal replicates from which detection data could be estimated. As previously discussed, these detection data were then analyzed using hierarchical models. Given the extra effort employed in the present work to collect replicate data to account for detection and the added complexity inherent in the proposed hierarchical modeling needed to analyze these data, we may ask what the specific benefits of accounting for the imperfect detection of heteropteran bug species were. I believe that our results suggest that our studies benefited from accounting for detectability in at least five important ways. First, because our estimates of species occupancy were corrected for detection bias, we were able to report reliable community-level comparisons between study treatments. These became especially important, for example, in *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* case study, where we were interested in comparing the species richness of two different habitat treatments within an urban green space. Second, we were able to report accurately the effects of covariates on occupancy. For example, not accounting for detectability in *Effects of urbanization on occupancy and species richness* may have led to underestimations of the negative effect of urbanization on heteropteran bug occupancy. Third, we were able to provide heteropteran bug species-specific (as well as guild, community, family and fauna-level) probabilities of detection. For example, results from all case studies indicate that heteropteran bugs show a low mean community-level (or fauna-level) probability of detection. These estimates may aid in understanding the ease or difficulty by which heteropteran bug species (and other levels) are observed in nature, which in turn may shed light on their rareness or commonality. Fourth, in *The effect of landscape functional heterogeneity on vineyard biodiversity* we used the detection estimates to identify the 'flight intercept' rather than the 'pitfall' trap as the more efficient method for surveying heteropteran bugs in vineyards. If this comparative analysis were be repeated with other insect taxa it could lead to the identification of the optimal sampling method or combination of methods for each taxonomical group and/or functional guild. Finally, we were able to hint at a potential relationship between heteropteran bug morphological and functional traits and their patterns of detection, as will be further discussed below in *Estimation of species and family detectability along macroecological gradients*. I note that the benefits of accounting for the observation process have been thoroughly demonstrated by other insect conservation-oriented studies (MacKenzie et al. 2005, Kéry et al. 2010b, van Strien et al. 2010), as well as studies preoccupied with the conservation of other animal taxa (MacKenzie et al. 2005, Russell et al. 2009, Zipkin et al. 2009, 2010, Martin et al. 2011)

To conclude, I wish to add some brief final observations on the challenging subject of scale. As Weins (1989) and Levin (1992) have pointed out, scaling issues are fundamental to both pure and applied ecological investigations. As previously mentioned, the case studies presented in this work were designed to cover at least three different scales of increasing spatial resolution and extent (landscape, shire and bioregion), which were considered (in theory but not in practice) to be hierarchically-nested. The latent goal of this arrangement was to grant us with means to compare how observed patterns such as species richness and occupancy varied across scales. Perhaps the most noteworthy result from this comparison is the observation that heteropteran bug mean community-level probabilities of detection and occupancy remained low and high, respectively, across the landscape and shire scales. Interestingly, because we specifically treated the species distribution data from the Iberian Peninsula bioregion as detection/non-detection rather than presence/absent data, the abovementioned pattern of low detection holds also for the bioregion scale. This findings seem to suggest that as a group heteropteran bugs are relatively common across the range of studied spatial extents and that their communities, as observed at the studied resolutions, include many difficult-to-detect rare species. It remains to be seen whether the pattern of high occupancy shown by heteropteran bugs at the landscape and shire scales will hold at bioregional or larger extents. A fine-grained (eg, resolved at the municipality or shire level) and well-replicated study could help to address this question.

5.2 Case studies

5.2.1 *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*

The general interest in understanding the role of green spaces in promoting biodiversity within urban landscapes has considerably increased the number of investigations focusing on golf courses and the habitats found within them (Frank and Shrewsbury 2004, Tanner and Gange 2005, Colding et al. 2009, Saarikivi et al. 2010). Here, we add an original contribution to this line of research by presenting an approach that explores the insect biodiversity benefits of novel grassland ecosystems within urban golf courses. An important general observation that may be made from our research is that urban golf courses present a highly diversified and trophically diverse heteropteran bug community. This is demonstrated by the 76 species detected in our study, which represent approximately 17% of the known Victoria heteropterofauna. Included among these species were economically-important predators such as the pacific damselbug *Nabis kinbergii* (Figure A2.17).

Our results suggest that novel grassland ecosystems within golf courses contribute to higher values of heteropteran bug species richness (Figure 4.18C). Our study also suggests that this pattern holds for both the herbivorous and predatory guilds (Figures 4.18A-B). Other similar studies have also highlighted the conservation potential of urban golf course for biodiversity. Studying odonates in permanent freshwater ponds within golf courses around Stockholm, Colding et al. (2009) found no difference between golf course and off-course ponds at the species, genus, or family levels. In golf courses within Helsinki, Saarikivi et al. (2010) showed that carabid beetles presented high levels of species richness' and that their assemblages were similar to those detected in the surrounding forest or farmland. Frank and Shrewsbury (2004), working in Maryland golf courses, installed beetle banks and flowering insectary stripes in the rough next to golf course fairways, and demonstrated that these 'conservation stripes' were successful in increasing insect predators and parasitoids. Our study, however, highlights the relevance for insect conservation of a novel ecosystem within urban golf courses that is sustained not by the reinforcement of management efforts, but rather through the cessation of them. Because these novel grasslands arose by secondary succession after the abandonment of management regimens over the 'rough' areas surrounding the golf course fairways, we believe them to be analogous to cropland 'oldfields' (Odum 1960) and would suggest they may be thought of as 'oldroughs'. The large number of heteropteran bugs species that we observed exclusively in oldroughs (over 40%), including the predators *Cermatulus nasalis* and *Coranus callosus*, an undescribed Antillocorini, and the alydid *Melanacanthus scutellaris*, which represents the first record of this species in Victoria, agrees with the findings of Small et al. (2003), Kattwinkel et al. (2011) and Robinson and Lundholm (2012) who reported that unmanaged urban habitats supporting ruderal or spontaneous vegetation may be rich depositories of scarce, rare and endangered insect biodiversity.

In this case study, we have analyzed species richness and occupancy with a special type of hierarchical linear model: the multi-species site occupancy model. One important strength of our approach is that we can estimate community-level attributes, such as species richness and mean community occupancy, as a function of the species-specific probabilities of occurrence. Moreover, we can refine these estimates by incorporating covariates to assess the effects of environmental factors on occupancy. For example, our modeling approach allowed quantification of the relationships between golf course vegetation structural complexity and heteropteran bug species richness and occupancy. For whole community species richness, we found that heteropteran bugs were positively associated with vegetation structure (Figure 4.20C). This positive relationship was equally identified for the herbivorous and predatory guilds (Figures 4.20A-B). We also found that there is a strong positive effect of vegetation structure on the mean community occupancy of heteropteran bugs (Figure 4.21). In fact, the posterior credible interval for this community hyperparameter contained only positive values. Thus, as our predictions of the effect of vegetation density on the species-specific probabilities of occurrence illustrate (Figure 4.22), heteropteran bug species in golf courses are, generally, much more likely to occur in patches of dense and complexly-structured vegetation than in those showing scarce and unstructured vegetation. This supports the results of other studies that have documented the positive effects of vegetation structure complexity on insect species richness (Lawton 1983, Dennis et al. 1998), including some that have illustrated the response of heteropteran bugs (Southwood et al. 1979, Kőrösi et al. 2012). The positive relationship between complexly-structured vegetation and species richness may be explained by the heterogeneity of microhabitats and diversity of resources that structurally complex vegetation habitats offer to insects (MacArthur 1972, Joern and Laws 2013). Our species richness results do not contradict this generalization. Species-specific responses, however, are not completely homogenous. Of the total of 76 species for which we estimated species-specific effects on occupancy, 14 (Figure 4.22A), including the species with the highest occurrence probabilities (the herbivores *Nysius caledoniae*, *Remaudiereana inornata* (Figure A2.26 bottom), *Mutusca brevicornis* (Figure A2.22), *Chaetedus longiceps* and the predator *Nabis kinbergii* (Figure A2.17)) were not influenced by vegetation structure. Thus, our results indicate that at least some species may be unaffected by general conservation-oriented management actions within golf courses, thereby highlighting the importance of incorporating species-specific responses into biodiversity assessments.

Many investigators have extensively discussed the challenges imposed by the imperfect detection of species (Gu and Swihart 2004, MacKenzie et al. 2006, Royle and Dorazio 2008, Kéry 2010, 2011). Surprisingly, issues surrounding detectability are still frequently overlooked by many ecological, biodiversity and conservation studies. An important strength of our research is that we specifically accounted for imperfect detection both in our sampling and modeling methodology. Thus, we are confident that our models have yielded true species richness and occupancy estimates, and unbiased covariate relationships.

Overall, our results demonstrate the conservation potential of unmanaged herbaceous vegetation and of complexly-structured vegetation patches in urban green spaces for insect biodiversity. In order to preserve and/or increase insect biological diversity in urban landscapes, special attention should be given to novel grassland ecosystems within large urban parks, especially since the availability of a rich insect fauna is important for other animal taxa such as birds and lizards. This might imply a change of management paradigm to one that deliberately incorporates areas of vegetation that are less formally managed, while striving to promote the structural complexity of the vegetation that is already under management.

5.2.2 *The effect of landscape functional heterogeneity on vineyard biodiversity*

The establishment of the European Union LIFE+ 2009 project 'Demonstrating biodiversity in viticulture landscapes' (BioDiVine) and of other research initiatives concerned with the conservation of biodiversity in vineyard regions (Thomson and Hoffmann 2010, Gillespie and Wratten 2012) have increased the interest in understanding how landscape heterogeneity may influence animal biodiversity, including the abundance, occupancy and species richness of beneficial predatory and parasitoid insects (Bianchi et al. 2006, Chaplin-Kramer 2011, Fahrig et al. 2011). In the present case study, we contribute new original insights into this active area of research by investigating the response of herbivorous and predatory heteropteran bugs to landscape functional heterogeneity. One interesting general finding of our study was the observation that large number of heteropteran bug species were present in the studied vineyards. In fact, the 149 heteropteran bug species detected in our study represent approximately 10% of the known Iberian Peninsula bioregion heteropterofauna. Our results also indicate that approximately 20% of these species were predators, including well-known natural enemies of crop pests such as *Orius laevigatus* and *Nabis pseudoferus* (Figure A3.1D).

Our vineyard sites were embedded in either a mosaic of simplified landscapes containing very few vegetation elements (Castellet i La Gornal) or a mosaic of complex landscapes containing a rich mix of natural or non-crop vegetation (Avinyó Nou). Our results suggest that on average the structurally complex Avinyó Nou wine sub-region sustained twice as much heteropteran bug herbivorous species as the simplified Castellet i La Gornal sub-region (Figure 4.24A). These findings agree with other studies that have reported community-level heteropteran bug response to landscape heterogeneity (Di Giulio et al. 2001, Kőrösi et al. 2012), as well as studies conducted with other insect taxa (Atauri and Lucio 2001, Marini et al. 2008). Our results also suggest that on average the simplified Castellet i La Gornal wine sub-region sustained an equal number of predatory species as the structurally complex Avinyó Nou sub-region (Figure 4.24B). This finding seems to contradict many studies that indicate a positive response of the species richness of natural enemies to landscape heterogeneity (reviewed in Bianchi et al. 2006 and Chaplin-Kramer et al. 2011). However, a closer look at the species composition of both sub-regions indicates that as much as 40% of all predators detected in our study were unique to the more complex Avinyó Nou sub-region. We believe that this highlights how community-level measures of diversity may mask the species-specific contributions to the effect under study, an issue that can be exacerbated if community estimates are based on raw counts and are not corrected for imperfect detection.

One original contribution of the present study is that we developed a measure of functional landscape heterogeneity (ie, proportion of natural habitat) that was specifically designed to account for the resource requirements of heteropteran bugs in viticulture landscapes, and incorporated it into our hierarchical linear models as a predictor of heteropteran bug community and species-specific occupancy. The mean estimate for the community-level effect of the proportion of natural habitat on heteropteran bug herbivorous species was positive, and its credible interval contained only positive values. This suggests that many herbivores in the community are more likely to occur as the proportion of natural habitat increases in the surrounding landscape. Not surprisingly, our species-specific predictions show that 73% of all herbivorous species experienced a slight to large increase (greater than 1.15 to

12-fold) in their probabilities of occurrence along the gradient of proportion of natural habitat (Figures 4.26E, G, I). Overall, these community-level and species-specific findings are consistent with our previous findings regarding species richness. Interestingly, nine species, including well-known polyphagous crop pests such as *Lygus pratensis*, *Oxycarenus lavaterae* and *Nezara viridula*, showed a negative response to proportion of natural habitat (Figure 4.26A). A possible explanation for this response might be that at low levels of proportion of natural habitat these species become abundant via a release from the predatory pressures that they are exposed to in more structurally-complex landscapes.

Although our models shows that the mean estimate for the community-level effect of proportion of natural habitat on heteropteran bug predatory species was positive, the credible interval for this community-level hyperparameter contained both positive and negative values. This finding suggests that on average the occupancy of predatory species increases concomitantly with the proportion of natural habitat present in the landscape, but this response is associated with a high degree of uncertainty that reflects the large variability in the species-specific responses. In fact, our results show that more than 80% of all predators experienced either a large increase (greater than 4-fold) or an equally large decrease in their probabilities of occurrence along the proportion of natural habitat gradient (Figures 4.26B, D, F, H, J). For example, our predictions show that the well-known natural enemy *Orius laevigatus* strongly responded to the increased proportion of natural habitat with decreased probabilities of occupancy (Figure 4.26D). By contrast, the assassin bug *Rhynocoris cuspidatus* (Figure 4.23) shows the opposite trend, a sharp increase in occupancy with increased proportion of natural habitat (Figure 4.26J). Again, this highlights the importance of considering detection-corrected species-specific responses as an alternative to raw counts of species richness in studies concerned with the preservation of biodiversity (Bennett et al. 2006, Russell et al. 2009, Zipkin et al. 2009).

Several mechanisms may explain the higher occupancy of most heteropteran bug herbivorous and predatory species on vineyards surrounded by a larger proportion of natural habitat. Most importantly, the diverse number of vegetation communities present in natural habitats may provide heteropteran bugs with a larger availability of living habitats (Bennett et al. 2006), including ‘keystone structures’ crucial for specialist species (Tews et al. 2004). Many studies have proposed (Odum 1960, Hobbs et al. 2006), among them a case study presented in this thesis (*The insect biodiversity benefits of novel grassland ecosystems in urban green spaces*), that novel ecosystems are rich repositories of biodiversity. For these reasons, we believe that field margins and oldfields may be considered keystone elements in viticulture landscapes. Therefore, we suggest that these habitats may be given special consideration in biodiversity conservation efforts in vineyard environments. Other complementary explanations state that natural habitats within agricultural landscapes have the potential to provide (Bennett et al. 2006, Bianchi et al. 2006): (1) extra resources (eg, nectar and pollen) essential to the feeding patterns of some species, (2) spatial structures necessary for some species to complete their life cycles (eg, hibernate), and (3) a moderate microclimate that may contribute to increase their biological fitness.

To conclude, this study clearly demonstrates that, regardless of the surrounding landscape, the flight intercept traps detect herbivorous and predatory heteropteran bugs more efficiently than the pitfall traps. We therefore suggest that future research focusing on this group may consider the flight intercept trap to be an effective surveying methodology. Of course, before a generalization can be conclusively made, this result should be further tested with other taxa to understand how different insect groups may be influenced by sampling methodology.

5.2.3 *Effects of urbanization on occupancy and species richness*

The ongoing interest in understanding the effects of urbanization on biodiversity (Hahs et al. 2009, Luck and Smallbone 2010, Pickett et al. 2011), including that of insects and other arthropods (McIntyre 2000, McIntyre and Rango 2009, Kotze et al. 2011) has led to a proliferation of studies aimed at assessing community-level responses to urbanization (McKinney 2008 and reference therein, Niemelä and Kotze 2009, Sattler et al. 2010, Price et al. 2011). Here, we propose an original addition to this research area by presenting an approach that simultaneously assesses heteropteran bug community-level and species-specific responses to a gradient of urbanization. A strength of our quantitative approach is that our species survey was coupled with a spatially and temporally replicated faunistic research of the study region (see *Heteroptera from El Maresme*) that provided the data to generate detection histories necessary for our hierarchical models to account for the imperfect detection of species. Another strength of our approach was that the study was not limited to responses along the urban-to-rural gradient of a single urban area. Instead, using a broad measure of urbanization we developed an urbanization gradient for the whole study region by quantifying the degree of urbanization of each one of its 30 municipalities. Thus, our gradient characterizes the whole spectrum of urban development to which the study area has been historically exposed using a measure that ensures the comparability and repeatability of the study across other urban regions. Finally, our study strongly benefited from the insect group we decided to work with. As our results highlight, heteropteran bugs proved to be interestingly diverse, both taxonomically (142 different species) and functionally (112 herbivores and 30 predators), while also presenting economically-important species, such as the pest natural enemy *Orius laevigatus laevigatus*.

Our results suggest that total heteropteran bug species richness and heteropteran bug herbivore and predatory species richness decreased along a gradient of increasing urbanization (Figure 4.29A-C). These results support the findings of other studies that have shown a negative response of species richness to urbanization. Reviewing results from 57 studies on different insect and arachnid taxa, McKinney (2008) reported that over 70% of the studies showed whole community species richness to peak at low levels of urbanization. Likewise, Niemelä and Kotze (2009), reviewing the response of carabid beetles to urban-to-rural gradients across eight cities, reported that, with a few exceptions, species richness decreased along the gradients. More recently, Sattler et al. (2010), working with at least 25 different insect and arachnid taxonomic groups, found that, while the species richness of herbivores showed no response, total species richness and the species richness of predators, including reduviid, nabid and anthocorid heteropteran bug species, responded negatively to urbanization.

The mean community-level effect of urbanization on the occurrence probability of heteropteran bugs was negative, and the posterior credible interval for this community hyperparameter contained only negative values. This suggests that most species in the community are more likely to occur as the degree of urbanization decreases along the gradient. Accordingly, approximately 97% of all herbivorous and 93% of all predatory species experienced on average a very large decrease (approximately a 8-fold change) in their probabilities of occurrence along the urbanization gradient (Figure 4.31B-E). These results therefore highlight that the heteropteran bug community of herbaceous ruderal vegetation in the study area are composed distinctly of 'urban avoider' species. Our results also indicate that a second group of species showed no response to the degree of urban disturbance (ie, on average they experienced a smaller than 1.09 change in their probability of occurrence along the urbanization gradient) (Figure 4.31A). Interestingly, these 'urban neutral' species included the polyphagous crop pest *O. lavaterae* (Figures 4.9 and A3.7B), which we have previously shown to have a positive response to human-induced perturbation in vineyards (see *The effect of landscape functional heterogeneity on vineyard biodiversity*), and the well-known polyphagous pest control agent *O. l. laevigatus* (Riudavets and Castañe 1998).

In this study of heteropteran bugs in the El Maresme shire, we have provided quantitative evidence for relationships between urbanization and species richness and occupancy. The nature of these relationships, however, remains correlative rather than causative, an issue that has been previously highlighted regarding the urbanization gradient approach in ecological and conservation investigations (Hahs et al. 2009). Nevertheless, as a first approximation to understand the causative mechanisms driving patterns of species richness and occupancy in urban regions, we believe our research may provide foundational knowledge essential for future studies. Our study, for example, focused exclusively on the heteropteran bugs living on urban habitat patches composed entirely of herbaceous ruderal vegetation. Thus, potential mechanistic explanations for our observed patterns may be better understood by researching the vegetation structural complexity of herbaceous ruderal urban ecosystems. In this respect, the case study *The insect biodiversity benefits of novel grassland ecosystems in urban green spaces* illustrated the positive effects of complexly-structured ruderal vegetation on heteropteran bug species richness and occupancy. We therefore anticipate that further studies may benefit from characterizing the urban context through more specific measures of urbanization (McDonnell and Hahs 2008, Hahs et al. 2009, Kotze et al. 2009). The gradient could be characterized, for example, with respect to the vegetation density of urban herbaceous ruderal patches. This approach may contribute to the elucidation of the fine-scaled mechanisms operating within urban ecosystem responsible for shaping insect communities and determining their patterns of occupancy. Moreover, they may advance our understanding of the role unmanaged herbaceous vegetation may play in the conservation of biodiversity in our ever-increasing urbanized world.

5.2.4 *Estimation of species and family detectability along macroecological gradients*

Detectability is an important source of stochasticity. By reason of its critical role in accounting for the uncertainty inherent to the processes of observation and measurement of large scale species distribution data, issues associated to the imperfect detection of species have been recognized as prominent areas of research (Whittaker et al. 2005, Richardson and Whittaker 2010, Kéry et al. 2010b, Kéry 2011, Beck et al. 2012). It is for this reason that I investigated the stochasticity associated with the observation process driving large scale patterns of heteropteran bug occupancy in the Iberian Peninsula bioregion and assessed the effects of macroecological gradients on heteropteran bug detection patterns. I want to stress that a pivotal strength of the research presented here is that it was explicitly linked with the development of the complementary faunistic study *Catalog of the Heteroptera from the Iberian Peninsula*, which provided distributional data for the Iberian Peninsula bioregion's 1,253 species and subspecies analyzed in this work. By coupling these two studies, I was able to simultaneously address large scale faunistic (eg, the 'Wallace shortfall') and macroecological (eg, imperfect detection) issues to the benefit of both pure and conservation science.

Although the Iberian Peninsula dataset used in this case study aggregated bibliographical, photographical and new field data gathered during almost 200 years of faunistic investigations, results suggest that on average the probability of detecting heteropteran bug species in the bioregion remains low (approximately 9%). Given the established relationships between detection and abundance/occupancy (Gaston and Blackburn 2000, Royle and Dorazio 2008), this finding therefore suggests that most heteropteran bug species in the Iberian Peninsula are either locally rare or present restricted range sizes. Another potential explanation, however, is that the low probability of detecting heteropteran bug species in the bioregion is considerably biased by the stochasticity associated with the sampling methodologies employed to detect the species (Dorazio 2007, Kéry and Schmidt (2008). This may be best highlighted by the striking asymmetries in the intensities by which districts and provinces in the Iberian Peninsula have been historically surveyed for heteropteran bugs. For example, the *Catalog of the Heteroptera from the Iberian Peninsula* dataset indicates that only 13 species have been detected in the 6,675 km^2 Portuguese district of Castelo Branco, which seems highly unlikely given the documented high levels of heteropteran bug diversity present in the bioregion.

Results also suggest that the probability of detecting a heteropteran bug varied considerably depending on the family the species belongs to. These family-level variations were consistent with biological attributes characterizing these taxa. Families consisting of large and robust species and genera (eg, Coreidae, Nepidae and Alydidae) presented on average higher detection probabilities. Other families showing above average detection probabilities were those known for the aposematic coloration displayed by most of their species, for example the Pyrrhocoridae and most Lygaeidae. On the other hand, families characterized by small and slender taxa (eg, Tingidae, Anthocoridae, Microphysidae and most Miridae) presented much lower than average probabilities of detection, as did families known by their very specific life-history traits, such as the burrowing Cydnidae and cryptic mycophagous Aradidae. Furthermore, results also suggest that the probability of detecting a heteropteran bug in the Iberian Peninsula varied distinctly from species to species. Species presenting the highest detection probabilities corresponded to species that were at least 8.5 mm in length (eg, the coreid *C. m. marginatus,* Figure 4.4) and conspicuously (eg, the pentatomid *C. fuscispinus,* Figure 3.4) and/or aposematically-colored (eg, the pentatomid *E. ornata* and lygaeid *S. pandurus,* Figures 4.3 and 4.6, respectively). A notable exception to this pattern, however, was the small and dull-colored lacebug *Dictyla echii* (Tingidae). Of special interest was the high probability of detection showed by the coreid *Leptoglossus occidentalis* Heidemann, 1910 (Figure 5.3), an alien Nearctic species recently established in Europe (Rabitsch 2010) that until 2003 had not been recorded in the Iberian Peninsula bioregion (Pérez and Prieto 2010). These findings hint at a potential relationship between the morphological and functional traits of heteropteran bugs and their patterns of detection/occupancy. As demonstrated by Pollock et al. (2012) and Palma et al. (*unpublished manuscript*), further insights into the mechanisms driving the distribution of species may be gained by quantifying how species traits modulate their response to the environment.

In this study, area, altitudinal range, mean annual temperature and population density had a positive effect on the mean heteropteran bug species detection probability (Figures 4.32A-C, E). Because this mean detection was estimated using family-level hyperparameters, this result suggests that the probability of detecting most Iberian Peninsula heteropteran bug families increases along the gradients generated by these macroecological variables (Figures 4.33A-C, E). Correspondingly, over 75% of all families experienced on average a greater than 8-fold increase in their detection probabilities from the low to high extremes of the gradients (Figures 4.34C-D, 4.35A-B, 4.36C-E and 4.38A-C). On the contrary, mean annual precipitation showed a negative effect on the bioregional-level probability of detection (Figure 4.32D), suggesting that the detection probability of most families decreases in response to increasing precipitation (Figure 4.33D). In fact, all families experienced a greater than 3-fold decrease along this gradient (Figure 4.37A-C). These findings indicate

Figure 5.3 The western conifer seed bug *Leptoglossus occidentalis* Heidemann, 1910. Source: Laurence Livermore (Flickr)

that either heteropteran bug abundance and occupancy are in fact influenced by the gradients or that the gradients have historically biased the rate at which heteropteran bugs have been detected. Some of these findings seem to agree with well-established macroecological patterns such as the 'species-area' relationship (Gaston and Blackburn 2000) and the spatial congruence between people and biodiversity (Luck 2007). However, a closer look at the data indicates important inconsistencies. For example, in the largest province in the Iberian Peninsula (Badajoz) only 2% of the total heteropterofauna has been historically detected. Incidentally, this province is amongst the least surveyed spatial units in the bioregion. I therefore believe that understanding the true response of species to macroecological gradients may require a more comprehensive account of the bias introduced by imperfect detection.

Many authors have previously highlighted the importance of detectability for inferring the 'true' rather than the 'apparent' distributions of species (Dorazio 2007, Kéry and Schmidt 2008, Royle and Dorazio 2008, Kéry et al. 2010b, Beck et al. 2012). Results presented in this case study further emphasize the need to account for imperfect detection in macroecological and conservation biogeography studies. I anticipate that methodologies that explicitly account for the observation process may prove fundamental in disentangling which components of species distributions at large bioregional scales are a consequence of imperfect detection as opposed to true patterns of occupancy.

6 Conclusions

1. Heteropteran bugs make an important contribution to the hyperdiversity of insects. As with other insect taxa, the proper species identification of heteropteran bugs requires diagnostic dichotomous keys. Faunistic studies can considerably benefit from the use of in-situ photographic records and biodiversity web resources, these two new taxonomical tools have the potential to expedite conservation-oriented research and engage the general public in the conservation of nature.

2. Faunistic syntheses such as catalogs and datasets are essential for documenting where species occur and how they are distributed, and may effectively contribute to overcome the 'Wallace shortfall' associated with large scale distributional data. The establishment of a research-oriented 'Heteropteran Bug Monitoring Scheme' may prove to be an effective faunistic tool that guarantees the constant flow of fine-grain, high-value species data. The gap between taxonomy and conservation may be bridged by explicitly coupling faunistic survey efforts to the monitoring of state variables relevant to ecological research.

3. The hierarchical view is an approach to quantitative ecology with the potential to simultaneously account for the stochasticity associated with the ecological and observation processes. Multi-species site occupancy models are effective quantitative tools that estimate species-specific probabilities of detection and occurrence, from which the size of the community (ie, species richness) may also be estimated. One important advantage of multi-species site occupancy models is the ease by which covariates are specified into the linear predictors of both the ecological and observation process levels.

4. Bayesian methods are powerful inferential tools for the conduction of quantitative research. Estimation of a given parameter under a Bayesian approach results in a posterior probability distribution that provides not only the mean but most importantly its associated uncertainty. The Bayesian mode of inference allows researchers to focus on effect sizes rather than statistical significance. Results from Bayesian analyzes can be communicated clearly and effectively to conservation policy-makers.

5. Species are imperfectly detected. The methods we use to survey insects are important sources of uncertainty that must be taken into account when studying their patterns of occupancy and species richness. Survey protocols should be specifically designed to include spatial and/or temporal replicates from which detection data can be estimated.

6. Heteropteran bugs, as a group, are relatively common across spatial extents, and their communities, as observed at fine grains of resolution, include many rare species that are difficult to detect. The assumption that this pattern holds at bioregional or even larger spatial extents requires more empirical investigation. A properly replicated study, resolved at the municipality or shire level, could contribute to address this gap in knowledge.

7. Novel grassland ecosystems supporting ruderal or spontaneous vegetation are rich depositories of heteropteran bug biodiversity. In urban landscapes, oldroughs and unmanaged urban herbaceous margins may play an important role in the conservation of heteropteran bug species and other insect taxa. Within urban green areas such as golf courses, the complexity of the vegetation structure has a positive effect on heteropteran bug species richness and the species-specific occurrence probabilities of most species.

8. Heteropteran bug species respond to the surrounding landscape. In viticulture landscapes, the proportion of natural habitat surrounding vineyards has a positive effect on mean herbivorous guild-level probability of occurrence. This positive effect is more uncertain for the predatory-guild. In the urban landscapes, species richness of both herbivores and predators decreased along a gradient of increasing urbanization. Almost all species studied were 'urban avoiders', experiencing on average very large decreases in their probabilities of occurrence along the urbanization gradient.

9. Heteropteran bugs in the Iberian Peninsula have a low probability of being detected. This probability varies markedly from family to family and from species to species. Macroecological variables have positive (area, altitudinal range, mean annual temperature and population density) or negative (mean annual precipitation) effects on the mean heteropterofauna detection probability. Methodologies that explicitly account for the observation process may prove fundamental for disentangling which components of species distributions at large bioregional scales are a consequence of imperfect detection as opposed to true patterns of occupancy.

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Resumen

Introducción y objetivos

Hasta el momento, el número de especies descritas supera el millón de taxones. Cabe destacar que la gran mayoría de estas especies corresponden a insectos, grupo que se caracteriza por su elevada diversidad a nivel planetario. Dicha diversidad representa una enorme funcionalidad que permite procesos como el reciclaje de nutrientes, la polinización y dispersión vegetal, la estructuración del suelo, etc. Los insectos juegan un papel extremadamente importante en mantener la salud de nuestro planeta y es por ello que necesitamos conservarlos, con el fin de que las funciones que realizan dentro de los ecosistemas no desaparezcan.

La conservación de las especies pasa obligatoriamente por su identificación, ya que sólo podremos proteger aquello que conocemos. Para ello, disponemos de la taxonomía, ciencia que nos permite clasificar los individuos dentro de una jerarquía biológica bien organizada. En un grupo tan diverso y escasamente estudiado como los insectos, es probable que sólo un pequeño porcentaje de las especies hayan sido descritas y que las medidas de conservación desarrolladas para mantener su complejidad funcional hayan sido escasas. Uno de los objetivos asociados a esta tesis consiste en unir conocimientos faunísticos (parte taxonómica) y ecológicos (parte cuantitativa) para entender mejor la biodiversidad que nos interesa preservar, a la vez que podamos tomar decisiones de gestión y desarrollar políticas de conservación basadas en resultados estadísticamente robustos. El primer paso para cumplir este objetivo pasa por la identificación hasta el nivel de especie de todo el material entomológico recogido en el campo. Para facilitar este proceso se desarrollaron 157 claves dicotómicas regionales de identificación. Por otro lado, se desarrollaron varios catálogos y bases de datos faunísticas, básicos para cualquier estudio de distribución de especies. Por último, la tesis incorpora fotografías in-situ y recursos electrónicos como registros de biodiversidad, ya que pensamos que son herramientas muy útiles para desarrollar la ciencia de la taxonomía a la vez que difunden la conservación de la naturaleza entre el público general, y lo hacen partícipe.

Esta tesis se limita al estudio de los insectos formalmente conocidos como Heteróptera Latreille, 1810. Estos representan un clado monofilético de insectos hemimetábolos, que en el contexto taxonómico se sitúan como una suborden del orden Hemíptera Linnaeus, 1758, y se dividen en siete clados o infraórdenes. El número estimado de especies de heterópteros descritas hasta ahora a nivel mundial es 42.347. Con esta tesis podemos situar el número de especies de heterópteros de la Península Ibérica en 1.453 (1.470 si incluimos subespecies). La monofilogenia de los heterópteros se basa en tres sinapomorfias morfológicas: i) apéndices de alimentación alargados, formando un rostro succionador-perforador que nace desde la parte delantera de la cabeza, ii) un par de glándulas odoríferas presentes en la pleura metatorácica de los adultos o en el dorso abdominal de estadios inmaduros, y iii) antenas de cuatro segmentos con dos escléritos intersegmentales. En cuanto a sus hábitos alimenticios, la mayoria son fitófagos, zoófagos (cabe destacar su importancia como controladores de plagas) o hematófagos. Además, estos insectos representan un grupo eurioico, pues utilizan con éxito una gran variedad de hábitats, lo que explica su distribución cosmopolita.

A lo largo de los diferentes casos de estudio de esta tesis, relacionamos las especies de heterópteros con datos ambientales mediante el uso de modelos cuantitativos. Así es como conseguimos realizar estudios ecológicos, que nos permiten conocer mejor la abundancia, distribución y diversidad de insectos a través de las escalas jerárquicas de la organización biológica. Un modelo es una representación matemática de las relaciones que nosotros creemos que existen entre varios elementos de un sistema estocástico. Estos modelos estadísticos representan una buena herramienta científica para describir y analizar datos procedentes de sistemas como el medio natural, con una incertidumbre asociada. La observación de la naturaleza genera, frecuentemente, datos organizados jerárquicamente. Trabajar con estos datos exige tener en cuenta esta organización y es por esta razón que a lo largo de la tesis utilizaremos modelos jerárquicos lineales (MJL). Los MJL son una generalización de los métodos de regresión basados en las relaciones entre distintas variables dentro de un conjunto de datos organizados jerárquicamente. Estos modelos pueden aplicarse con éxito a sistemas complejos, en los que la estocasticidad actúa a varios niveles, e incluyen variables aleatorias e hiperparámetros. La estimación de los parámetros se realiza mediante inferencia bayesiana, cuyo principio básico es el conocido Teorema de Bayes. Dicho teorema proporciona la distribución posterior del parámetro a estimar, o lo que es lo mismo, la distribución de probabilidad de obtener dicho parámetro dados los datos recogidos en el campo y la información que previamente conocíamos sobre el parámetro. En otras palabras, nuestros modelos estadísticos combinan el conocimiento previo con nuestros nuevos datos para generar nuevo conocimiento (o conocimiento posterior). Uno de los puntos fuertes de la inferencia bayesiana es que nos permiten estimar la probabilidad de que nuestras hipótesis sean verdaderas, a diferencia de los métodos de inferencia frecuentista que trabajan con p-valores. Además, este tipo de inferencia se centra en analizar el denominado "tamaño del efecto" y los intervalos de credibilidad (precisión con que se estiman los parámetros), que dan una idea del poder estadístico con el que estamos trabajando.

A lo largo de nuestros casos de estudio los modelos jerárquicos lineales no son más que una herramienta para observar, medir y analizar cómo varían nuestras variables respuesta con respecto a distintas variables ambientales explicativas. Así es como intentamos establecer interacciones y causas para los procesos ecológicos que se observan, sin olvidar que siempre están acompañados de su incertidumbre asociada. Ya que nuestros intereses de investigación se centran en cuestiones relacionadas con la distribución y diversidad de especies, y por tanto con los niveles de especie y comunidad de los sistemas ecológicos, las variables respuesta con las que trabajamos son ocurrencia y riqueza específica. La ocurrencia se define como el número o proporción de unidades espaciales en las que una especie habita, y la riqueza específica es el número de especies que habitan en cada una de esas unidades.

En cuanto a la incertidumbre asociada a los procesos ecológicos, ésta puede tener dos orígenes. El primero es la variación espacial. Para evitar errores ligados a la misma, nuestro diseño del trabajo de campo incluye una o más fuentes de aleatoriedad y estratificación en la elección de las unidades espaciales. La otra fuente de incertidumbre está ligada a la detectabilidad. La detectabilidad es la capacidad del observador de detectar el 100% de los organismos que está buscando en un lugar. Generalmente, la detectablidad es imperfecta y, como consecuencia, nuestras observaciones no son un fiel reflejo de lo que realmente hay en el lugar de estudio. Por tanto, trabajar sin tener en cuenta una detección imperfecta puede llevarnos a resultados y conclusiones poco realistas. Nuestros datos son analizados mediante modelos estadísticos precisamente desarrollados para incluir la detectabilidad. Por otro lado, los patrones biológicos vienen definidos por diferentes niveles ecológicos y no incluir esta información también podría generar resultados sesgados. Por esta razón, la tesis intenta también tratar temas de escala, explorando y comparando cómo dichos patrones varían bajo diferentes escalas.

Incluimos cuatro casos de estudio de naturaleza cuantitativa, investigaciones originales del autor y colaboradores realizadas entre 2010 y 2013. Todos ellos se enfrentan a cuestiones ecológicas, de biodiversidad y conservación, compartiendo los siguientes puntos: i) fuerte base taxonómica de los organismos modelo, los insectos heterópteros, ii) cuantificación de los sistemas estocásticos bajo estudio, incluyendo múltiple causalidad e incertidumbre, y iii) uso de modelos jerárquicos lineales e inferencia bayesiana. A continuación se listan los casos de estudio incluidos en la tesis:

1 Beneficios de los ecosistemas herbáceos noveles de espacios verdes urbanos sobre la biodiversidad de insectos

2 Efecto de la heterogeneidad funcional del paisaje sobre la biodiversidad de los viñedos

3 Efecto de la urbanización sobre la ocupancia y la riqueza específica

4 Estimación de la detectabilidad de especies y familias a lo largo de gradientes macroecológicos

El primer caso de estudio nace a partir de la participación del autor en el proyecto "Servicios ecológicos de grandes espacios verdes urbanos – los beneficios de los campos de golf urbanos sobre la biodiversidad y el carbono", dirigido por Stephen Livesley y financiado por el Consejo Nacional de Investigaciones Australiano (ARC). La existencia de nuevos ecosistemas, en particular hábitats herbáceos noveles, que nacen como consecuencia de la actividad humana fue el punto en partida de este análisis. Estos hábitats noveles se caracterizan por combinaciones de especies noveles, con potencial para modificar el funcionamiento y las características de los ecosistemas. De hecho, nuestro estudio se basa en analizar si la existencia de estos nuevos ecosistemas modifican la biodiversidad de insectos dentro de los campos de golf. Más concretamente, queremos saber si su presencia favorece el aumento de riqueza específica ligada a estructuras vegetales que tradicionalmente aparecen en los campos de golf, los bosques. Para ir un paso más alla, también cuantificamos la relación entre dicha biodiversidad y la complejidad estructural de la vegetación en dichos campos de golf. Este caso de estudio se desarrolla en el el sudeste de Melbourne, Victoria, Australia.

El segundo caso de estudio se enmarca dentro de un proyecto más amplio denominado "Evidenciando la biodiversidad de los paisajes vitivinícolas". Se trata de un proyecto LIFE+, desarrollado por la Unión Europea y coordinado por Joël Rochard, que comenzó en el año 2009. El objetivo de nuestro análisis es análizar la influencia de la complejidad del paisaje en la diversidad de insectos. Para ello, se desarrolla un índice de heterogeneidad de paisaje funcional, que se incorpora en los modelos como variables explicativa. Por otro lado, el uso de diversos métodos de captura de insectos durante el proyecto resulta una herramienta muy útil a la hora de estudiar la detectabilidad de los mismos en los viñedos. Este caso de estudio se desarrolla en la comarca del Alto Penedés, Cataluña, Peninsula Ibérica.

Los dos últimos casos de estudio pretenden ser una muestra de cómo los datos derivados del monitoreo de especies y el trabajo de campo pueden usarse simultáneamente para responder cuestiones faunísticas y ecológicas. En cuanto al tercer caso de estudio, el objetivo es inferir el efecto de la urbanización sobre la ocupancia y la riqueza específica de los insectos que aparecen en hábitats herbáceos ruderales. En este caso la variable cuantitativa que se utiliza para explicar la respuesta biológica es el Índice de Week, que integra información demográfica y de paisaje para sintetizar el grado de urbanización. Este análisis se desarrolla en la comarca del Maresme, Cataluña, Península Ibérica.

El último caso de estudio se centra en investigar patrones de ocupancia y distribución de especies a nivel de toda una bioregión, la Península Ibérica, incluyendo Portugal, España y Andorra. Como primer paso para este estudio, se desarrolla el *Catálogo de los Heterópteros de la Península Ibérica*, que recoge información sobre la distribución de 1470 especies y subespecies a lo largo de dicha área. En una segunda fase, se intentan explicar los patrones de distribución anteriormente recogidos en el Catálogo en base a variables cuantitativas. Mediante modelos jerárquicos se definen la probabilidad de detección específica de cada especie, la probabilidad de detección específica de cada familia y la probabilidad de detección del conjunto de heterópterofauna a nivel de la Península Ibérica. Entonces se incluyen gradientes macroecológicos (rango altitudinal, temperatura media, precipitación annual, etc.) para explicar dichos patrones de distribucion de los insectos.

El objetivo principal de esta tesis es investigar la ecología, biodiversidad y conservación de los insectos heterópteros, mediante el uso conjunto de herramientas taxonómicas y de ecología cuantitativa. En cuanto a la taxonomía, quiero enfatizar la importancia de las claves de identificación, reconocer el papel de los catálogos y las bases de datos faunísticas para examinar patrones ecológicos, y aumentar el interés por los recursos fotográficos disponibles en la red. En lo que respecta a los análisis cuantitativos, muestro cómo los modelos jerárquicos lineales pueden usarse para estimar riqueza específica y ocupancia, así como para cuantificar los efectos de las covariantes ambientales que les influyen. Además, quiero demostrar el poder del método de inferencia bayesiano, las ventajas de la incorporación de la incertidumbre y el efecto de la escala en los patrones biológicos.

Métodología

La recolección de los heterópteros en el campo se llevó a cabo por varios medios; mangas entomológicas, aspiradores, trampas Berlese, trampas pitfall y trampas de intersección de vuelo. Los especímenes fueron preservados en etanol, de 70º e identificados hasta especie. Además del trabajo de campo realizado por el autor y colaboradores, se desarrollaron recursos electrónicos abiertas donde se inició un registro fotográfico de los heterópteros encontrados en la Península Ibérica y en Australia en la plataforma web Flickr. También se cedieron y tomaron registros fotográficos de distintas especies de heterópteros en las plataformas electrónicas Biodiversidad Virtual y The Encyclopedia of Life.

Tras establecer la nomenclatura taxonómica más actual y definir la sinonimia existente, se inició una búsqueda sistemática de literatura entomológica a fin de encontrar todas las citas de heterópteros existentes para la Península Ibérica y para Victoria, Australia. A este conjunto de citas históricas, se le sumaron las nuevas citas recogidas por el autor y colaboradores mayoritariamente a lo largo de la tesis, así como las citas en formato de fotografía digital disponibles gracias a las tres plataformas antes mencionadas. El conjunto de toda esta información permitió desarrollar los siguientes catálogos:

i. el *Catálogo de los Heterópteros de la Península Ibérica*, para el cual a cada registro se le asignó una unidad espacial concreta (a nivel de provincia en España, distrito en Portugal y país en Andorra).

ii. *Heterópteros de El Maresme*. En este caso las unidades espaciales corresponden a los municipios de la comarca.

iii. *Pyrrhocoridae de la Península Ibérica*. Mismas unidades espaciales que el Catálogo.

iv. *Heterópteros de Victoria*.

Por otro lado, la identificación de especímenes requiere de la existencia de claves de identificación desarrolladas para el grupo estudiado, actualizadas y accesibles. Durante esta tesis se desarrolaron claves dicotómicas de identificación válidas para la identificación de especímenes físicos y/o a partir de fotografía. Algunos ejemplos se listan a continuación:

i. *Clave para las familias de Heterópteros de Victoria*.

ii. *Clave para las familias de Heterópteros de la Península Ibérica*.

iii. *Clave para las tribus de Míridae de la Península Ibérica*.

iv. *Clave para los géneros de Rhyparochrominae de la Península Ibérica*.

v. *Clave para las especies de Deraeocoris de la Península Ibérica*.

Los métodos estadísticos que se utilizan en los análisis son, como ya se ha dicho con anterioridad, modelos jerárquicos lineales. Más concretamente, son una extensión de estos que permite trabajar simultáneamente con información relativa a un conjunto de especies e incorporar su detectabilidad. Estos modelos se denominan "modelos de ocupancia de lugar multiespecíficos (msSOM)" y nos permiten determinar la probabilidad de ocupancia y detección de cada especie individual, así como del conjunto de la comunidad de insectos. La inferencia estadística es, como también se ha comentado con anterioridad, de tipo bayesiano. Para ello, se especifica un conocimiento previo (priors) no informativo. Según lo más apropiado en cada caso, la distribución de probabilidad utilizada puede ser Normal, Uniforme, Bernoulli o Gamma. Dicha inferencia se basa en los algoritmos denominados Cadena de Markov – Monte Carlo, como vienen implementados en el software OpenBUGS. El estadístico de Gelman-Rubin se utiliza para evaluar la correcta convergencia de los parámetros estimados. Todos los análisis se llevan a cabo mediante los software R y OpenBUGS.

Para el primer caso de estudio, que analiza los beneficios de los ecosistemas herbáceos noveles de espacios verdes urbanos sobre la biodiversidad de insectos, las muestras se recogieron de ocho parcelas de 600 m^2 escogidas al azar en cada uno de los 13 campos de golf. Dichas parcelas pertenecían a una de las siguientes categorías; a) zona boscosa, o b) zona herbácea. La vegetación de cada parcela también fue muestreada. Un primer modelo de ocupancia de lugar multiespecífico (msSOM) incondicional se utiliza para estimar la riqueza específica total, de herbívoros y de predadores en la zona boscosa, así como las probabilidades de ocurrencia y detección. Después, se usa un segundo modelo similar pero esta vez para el conjunto de riqueza específica de la zona boscosa más la zona herbácea. Finalmente, se incluye una covariante relativa a la densidad de la vegetación para estimar su efecto en la riqueza de insectos y su probabilidad de ocurrencia.

En cuanto al efecto de la heterogeneidad funcional del paisaje sobre la biodiversidad de los viñedos, los datos se recogieron en 10 viñedos de una hectárea elegidos al azar dentro de dos subregiones con diferente heterogeneidad funcional del paisaje (simple y complejo respectivamente). Por cada viñedo se recogieron 20 réplicas, usando tanto trampas pitfall como de intercepción de vuelo. La riqueza específica de insectos herbívoros y predadores se modeló usando varios msSOMs que incluían como variables explicativas: el efecto de la subregión, el efecto del tipo de trampa para insectos y la influencia de la covariante referente a la complejidad del paisaje.

Para estudiar el efecto de la urbanización sobre la ocupancia y la riqueza específica de los heterópteros, las muestras se recogieron de dos parcelas ruderales seleccionadas al azar dentro del núcleo urbano de cada municipio. Cuando no fue posible encontrar dos parcelas ruderales, se muestreó al menos una. Cada parcela se visitó dos veces. El Índice de Week se calculó para cada núcleo urbano y se incluyó en un msSOM que permitió estimar la riqueza específica del total de heterópteros, así como de la fracción de herbívoros y predadores, y cuantificar el efecto de la covariante sobre las probabilidades de ocurrencia y detección de las especies.

Por último, la estimación de la detectabilidad de especies y familias a lo largo de gradientes macroecológicos requirió la utilización de los datos recogidos en el *Catálogo de los Heterópteros de la Península Ibérica*. Los gradientes macroecológicos hacen referencia a las siguientes covariantes; área, rango altitudinal, temperatura media annual, precipitación media annual y densidad de población. Un primer msSOM permitió estimar las probabilidades de detección de las especies, las familias y el conjunto de heterópterofauna de la Península Ibérica. Nuevos msSOM que incorporaron las cinco covariantes de forma independiente permitieron conocer el efecto de las mismas en las probabilidades calculadas con anterioridad.

Resultados

A lo largo de la tesis, se recogió un total de 512 especies y 16.855 especímenes de heterópteros. 418 especies fueron recolectadas en la Península Ibérica (representando aproximadamente el 30% de la diversidad conocida a día de hoy), mientras que en Victoria se recogieron 95 especies (22 %). Una única especie, *Nezara viridula*, se recolectó en ambas zonas.

Se generaron 4.180 nuevas citas, el 85.5% para la Península Ibérica y el resto para Victoria, Australia. La recolección se llevó a cabo en más de 400 localidades. Las 334 localidades correspondientes a la Península Ibérica engloban más de la mitad de las provincias y distritos incluidos en los trabajos de esta tesis. En cuanto a las 69 localidades muestreadas en Victoria todas ellas forman parte del área metropolitana de Melbourne. También se generaron 74 nuevos registros fotográficos, 68 para la Península Ibérica y 6 para Victoria, que se incluyen en los grupos de Flickr *Heterópteros de la Península Ibérica* y *Heterópteros de Australia*, respectivamente, y la mayoría también en el registro electrónico *Encyclopedia of Life*. Las especies con mayor número de registros fotográficos procedentes de la fuente electrónica *Biodiversidad Virtual* fueron *Carpocoris fuscispinus* y *Nezara viridula*. Gracias a esta base fotográfica se ha registrado la presencia de *Mecidea lindbergi*, especie nueva para la Península Ibérica.

El *Catálogo de los Heterópteros de la Península Ibérica* incluye 1470 especies y subespecies, 46 familias y 480 géneros. El mayor número de registros pertenece a la provincia de Barcelona y el menor al distrito de Castelo Branco. Esta tesis doctoral contribuye con 3.572 nuevos registros, pertenecientes a un total de 418 especies, para la Península Ibérica. La publicación del catálogo se realizará mediante la Sociedad Entomológica Aragonesa como parte de su serie de monográficos online.

El trabajo *Heterópteros de El Maresme* incluye 1.860 registros; el 58% procedente de nuevas citas de campo, el 40.7% de referencias bibliográficas y el 1.9% de registros en formato fotográfico. El catálogo engloba 323 especies pertenecientes a 33 familias. La familia mejor representada es la de los Miridae, seguida por los Pentatomidae, y la especies más ubicua *Eurydema oleracea*. El municipio con más registros fue Calella, mientras que Vilassar de Mar fue el que tuvo menor cantidad de citas.

El tercer trabajo, *Pyrrhocoridae de la Península Ibérica*, se centra en *Pyrrhocoris apterus* y *Scantius aegyptius*, las únicas dos especies pertenecientes a la familia Pyrrhocoridae que habitan en la Península Ibérica. Las referencias bibliográficas sitúan a la primera especie en el 42% del territorio ibérico y a la segunda en el 39%. Estos porcentajes aumentan hasta el 90 y el 51%, respectivamente, cuando incluimos información relativa a nuevas citas de campo y registros fotográficos. Este trabajo está enviado a la revista Heteropterus Revista de Entomología.

Por último, *Heterópteros de Victoria* incluye 438 especies y subespecies, 48 familias y 277 géneros. Pentatomidae es la familia mejor representada, que junto a Reduviidae y Miridae, engloban aproximadamente el 50% de las especies presentes en este estado australiano. Este trabajo incluye 746 nuevas citas de campo para Victoria, referentes a 95 especies de heterópteros, siendo tres de ellas primeras citas para dicha región.

Por otro lado, se generaron 157 claves dicotómicas de identificación; 2 a nivel general para las familias, 19 para familias concretas, 18 para subfamilias, 16 para tribus y 102 para géneros.

Con respecto a los casos de estudio, en el primero, que investiga los beneficios de los campos de golf urbanos sobre la biodiversidad de insectos, se recolectaron 6.628 heterópteros (22 familias, 67 especies de herbívoros y 9 de predadores). Los modelos permitieron hacer las siguientes estimaciones: i) la riqueza específica media de heterópteros de las zonas boscosas de los campos de golf es de 35 especies de heterópteros -30 herbívoros y 5 predadores-, ii) la riqueza específica media del conjunto de zonas boscosas y zonas herbáceas es de 60 especies -52 herbívoros y 8 predadores-, iii) la probabilidad de ocurrencia media es alta para ambos análisis mientras que la probabilidad de detección media es baja, y iv) el efecto de la densidad de la vegetación en la riqueza de heterópteros es positivo, aunque con grandes diferencias en la probabilidad de ocurrencia de distintas especies.

En el estudio sobre el efecto de la heterogeneidad funcional del paisaje sobre la biodiversidad de los viñedos se recolectaron 910 especímenes, pertenecientes a 149 especies de heterópteros (119 herbívoros y 30 predadores). Las estimaciones realizadas tras aplicar los modelos estadísticos a los datos fueron las siguientes: i) la riqueza específica media de la subregión con paisaje simple se sitúa en 48 especies de herbívoros y 13 de predadores, ii) el mismo dato para la zona de paisaje complejo es de 70 y 11, respectivamente, iii) la probabilidad de ocurrencia media para los herbívoros es más alta dentro del paisaje complejo, mientras que para los predadores es similar para los dos tipos de paisaje, iv) la probabilidad de detección media es baja, aunque mayor para trampas de intercepción de vuelo que para trampas pitfall, v) el efecto de la complejidad del paisaje sobre la diversidad fue positivo en la mayoria de los casos, aunque con grandes diferencias en la probabilidad de ocurrencia de distintas especies.

En el estudio que analiza el efecto de la urbanización sobre la ocupancia y la riqueza específica de los heterópteros, se recogieron 142 especies de heterópteros, siendo 112 herbívoras y 30 predadores. El modelaje de estos datos permitió estimar: i) la riqueza media de heterópteros en los municipios es de 108 especies -85 herbívoros y 23 predadores-, ii) la probabilidad de ocurrencia media es alta, mientras que la probabilidad media de detección es baja, iii) tanto la riqueza total de especies, como la cantidad de especies de herbívoros y de predadores, disminuye a lo largo del gradiente de urbanización, sin embargo su efecto en la probabilidad de ocurrencia de cada especie individual varía.

Por último, en cuanto a la estimación de la detectabilidad de especies y familias a lo largo de gradientes macroecológicos, los modelos arrojan los siguientes resultados: i) la probabilidad de detección media para los heterópteros a nivel de la Península Ibérica se estima aproximadamente en 0.1, siendo este valor variable en función de la familia, ii) las familias con mayor probabilidad de ser detectadas son Coreidae, Heterogastridae y Nepidae, iii) el efecto medio del área, el rango altitudinal, la temperatura media anual y la densidad poblacional sobre la probabilidad de detección de los heterópteros resulta positivo, mientras que el efecto medio de la precipitación media anual es negativo, iv) existen diferencias en el efecto de estas covariantes en la probabilidad de detección de cada familia.

Principales conclusiones

Los heterópteros contribuyen de forma importante a la hiperdiversidad de los insectos. La elaboración de claves dicotómicas actualizadas, ajustadas a la región de estudio y basadas en caracteres de fácil observación, facilita la correcta identificación de los ejemplares. Los estudios faunísticos pueden beneficiarse considerablemente de la existencia de registros fotográficos in-situ y recursos electrónicos de registro de la biodiversidad, ya que ambas herramientas taxonómicas tienen el potencial de favorecer la investigación dirigida a la conservación, así como de implicar al público general en dicho proceso.

Las síntesis, los catálogos y las bases de datos faunísticas son esenciales para registrar la ocurrencia y distribución de las especies. La utilización conjunta de estos recursos junto con variables ambientales relevantes para la ecología es clave para acortar la distancia que existe entre ciencias como la taxonomía y la conservación.

La visión jerárquica supone una aproximación a la ecología cuantitativa, que tiene el potencial de incluir la estocasticidad asociada a los procesos ecológicos y de observación humana de la naturaleza. Los “modelos de ocupancia de lugar multiespecíficos (msSOM)“ suponen herramientas quantitativas efectivas para estimar la probabilidades de detección y ocurrencia específicas de especies y lugares, así como el tamaño de la comunidad.

Los métodos bayesianos de inferencia estadística son herramientas poderosas para realizar análisis cuantitativos. La estimación de los parámetros viene asociada a su incertidumbre y al cálculo del “tamaño del efecto”. Y los resultados pueden comunicarse de forma clara y efectiva a los responsables de desarrollar políticas de conservación.

Los heterópteros son detectados de forma imperfecta. Una buena fuente de incertidumbre depende de los métodos que utilicemos para muestrearlos, por lo que debe tenerse en cuenta el proceso de observación a la hora de analizar patrones de ocupancia y riqueza específica. Los protocolos de muestreo deberían estar diseñados de forma que incluyan réplicas espaciales y/o temporales que permitan calcular la detectabilidad.

Los insectos heterópteros son relativamente comunes a lo largo de grandes áreas, y sus comunidades incluyen especies raras sólo detectables a pequeña escala. La asumpción de este patrón para la escala bioregional y superiores requiere mayor investigación.

Los sistemas herbáceos noveles con vegetación ruderal o espontánea contienen una gran diversidad de insectos heterópteros. Como parte del paisaje urbano, los márgenes herbáceos y otras formaciones poco o nada gestionadas representan zonas importantes para conservar la diversidad de heterópteros y otros insectos. En cuanto a zonas verdes urbanas, como los campos de golf, la complejidad en la estructura de la vegetación tiene un efecto positivo en la biodiversidad de insectos.

Los heterópteros se ven influenciados por el paisaje. En paisajes vitivinícolas, la ocurrencia de insectos herbívoros está favorecida por la proporción de hábitat natural favorable. En paisajes urbanos, la urbanizacion tiene un efecto negativo en la presencia de herbívoros y predadores. Ambos estudios, a partir de análisis distintos, convergen hacia una única conclusión.

A nivel de la Península Ibérica, la probabilidad de detectar heterópteros es baja y varía considerablemente entre familias y especies. Variables macroecológicas como el área, el rango altitudinal, la temperatura media anual y la densidad de población favorecen una mayor probabilidad de detección de heterópteros. Otras, como la precipitación media anual, tienen el efecto contrario. La utilización de metodologías que tienen en cuenta la incertidumbre asociada al proceso de observación es fundamental para distinguir qué componentes de la distribución de las especies a escala bioregional son producto de una detección imperfecta y cuáles son verdaderos patrones de ocupancia.

Appendix I Catalog of the Heteroptera from the Iberian Peninsula

Family Rhyparochromidae Amyot and Serville, 1843

On form and format

Occurrence of species in Andorra, Portugal and/or Spain is indicated by the three-letter abbreviations AND, POR and/or SPA, respectively. These abbreviations are followed by the full names in alphabetical order of the districts (POR) or provinces (SPA) where the species is documented to occur. The names of the spatial units may be accompanied by the following symbols:

> ! indicates a new field record for the district/province as documented in the present work.
> * indicates the species is documented in the given district/province exclusively by one or more photographic record(s).
> ** indicates the species is documented in the given district/province exclusively by a photographic records contributed to the Flickr group *Heteroptera from the Iberian Peninsula*.
> !** indicates that specimen that constituted the new record was also recorded photographically and contributed to the Flickr group *Heteroptera from the Iberian Peninsula*.
> ? indicates uncertainty about the species distribution in the give district/province as documented in the literature.

Subfamilies, tribes, genera and subgenera are arranged alphabetically within their appropriate higher taxon. *Synonymies and older name combinations* given under the header of the same name are limited to those encountered by us while researching the entomological literature used in the present work, no attempt was made to provide an exhaustive list for any given species.

New records have been formated using the following pattern:

District/Province: Locality 1, (nested within) Locality 2, date, person(s) responsable for field collection (leg.), person(s) responsable for species identification (det.), number and sex of specimens, sampling method (if availbale), habitat and/or host plant (if available), elevation (if available).

Within the list of new records the following symbols and abbreviations were used:

♀: female(s)
♂: male(s)
B: Brachypterous
LM: Luis Mata
m: meters
MG: Marta Goula
NP: National/Natural park

Suborder	**Heteroptera** Latreille, 1810
Infraorder	**Pentatomomorpha** Leston, Pendergrast and Southwood, 1954
Superfamily	**Lygaeoidea** Schilling, 1829
Family	**Rhyparochromidae** Amyot and Serville, 1843
Subfamily	**Plinthisinae** Slater and Sweet, 1961
Tribe	**Plinthisini** Slater and Sweet, 1961
Genus	*Plinthisus* Stephens, 1829
Subgenus	*Isioscytus* Horváth, 1876
1	*Plinthisus andalusicus* Wagner, 1963
Distribution	POR: Beja SPA: Cádiz Jaén Sevilla
References	Péricart (1998a, 2001) and Ribes J. (1974, 1986).
2	*Plinthisus minutissimus* Fieber, 1864
Distribution	POR: Castelo Branco SPA: Ávila Baleares Barcelona Cáceres Gerona Madrid Orense Pontevedra Toledo
References	Bator (1957), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Péricart (1998a, 2001), Ribes E. et al. (2000), Ribes J. (1990), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
3	*Plinthisus reyi* Puton, 1882
Synonymies	*Plinthisus bicolor* Rey, 1888
Distribution	POR: Coimbra SP: Albacete Ávila Baleares Cáceres Ciudad Real Cuenca Madrid Málaga
References	Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001) and Ribes J. (1965, 1990).
4	*Plinthisus saundersi* Horváth, 1893
Distribution	SPA: Cádiz Sevilla
References	Péricart (1998a, 2001).
Subgenus	*Nanoplinthisus* Wagner, 1963
5	*Plinthisus laevigatus* Puton, 1884
Distribution	SPA: Ciudad Real Córdoba Granada Madrid Murcia Sevilla
References	Péricart (1998a, 2001).
6	*Plinthisus magnieni* Péricart and Ribes J., 1994
Distribution	SPA: Alicante Barcelona Cáceres Castellón Ciudad Real Cuenca Murcia Tarragona Teruel Zaragoza
References	Péricart (1998a, 2001) and Ribes J. et al. (1997, 2004).
7	*Plinthisus megacephalus* Horváth, 1876
Distribution	POR: Coimbra SPA: Ávila Madrid
References	Costas (2004), Costas and Vázquez (2004) and Péricart (1998a, 2001).
8	*Plinthisus pilosellus* Horváth, 1876
Distribution	SPA: Cádiz
References	Péricart (1998a, 2001).
9	*Plinthisus pygmaeus* Horváth, 1882
Distribution	SPA: Cádiz
References	Péricart (1998a, 2001).
Subgenus	*Plinthisomus* Fieber, 1864
10	*Plinthisus pusillus* (Scholz, 1847)
Distribution	SPA: Ciudad Real Lérida Madrid
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).

Subgenus	*Plinthisus* Stephens, 1829
11	*Plinthisus brevipennis* (Latreille, 1807)
Synonymies	*Plinthisus autrani* Horváth, 1898
Distribution	AND POR: Bragança Coimbra Guarda Setúbal SPA: Albacete Ávila Barcelona Burgos Cáceres Cádiz Granada Lérida Madrid Pontevedra Salamanca Segovia Teruel Zaragoza
References	Bator (1957), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992, 2005), Péricart (1998a, 2001), Ribes E. et al. (2000), Ribes J. (1974), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004), Vázquez et al. (2003) and Wagner (1960a).
12	*Plinthisus convexus* Fieber, 1864
Distribution	POR?
References	Péricart (2001).
13	*Plinthisus flavipes* Fieber, 1861
Distribution	SPA?
References	Péricart (2001).
14	*Plinthisus jordiRibes J.i* Rieger & Pagola-Carte, 2011
Distribution	SPA: Murcia
References	Rieger and Pagola-Carte (2011).
15	*Plinthisus lepineyi* Vidal, 1940
Distribution	SPA: Córdoba
References	Péricart (1998a, 2001).
16	*Plinthisus longicollis* Fieber, 1861
Distribution	POR SPA: Albacete Ávila Badajoz Baleares Barcelona Cáceres Cádiz Ciudad Real Gerona Huelva Lugo Madrid Málaga Teruel Toledo Valladolid
References	Bator (1957), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Español (1964), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes E. et al. (2000), Ribes J. (1967, 1990), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New Records	Barcelona: Canyelles, Barcelona, 11/05/2010, LM leg., MG det., 1♀ 1♂, Berlese.
17	*Plinthisus major* Horváth, 1876
Distribution	POR: Portalegre SPA: Madrid
References	Péricart (1998a, 2001).
18	*Plinthisus putoni* Horváth, 1876
Distribution	POR: Faro Lisboa SPA: Albacete Baleares Barcelona Cádiz Ciudad Real Cuenca Jaén Lérida Madrid Málaga Tarragona!
References	Péricart (1998a, 2001), Ribes J. (1965, 1967), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New Records	Tarragona: Mont Caro, Els Ports de Beseit NP, 21/06/2012, LM leg. & det., 1♀, Herbaceous vegetation.
Family	**Rhyparochrominae** Amyot and Serville, 1843
Tribe	**Antillocorini** Ashlock, 1964
Genus	*Tropistethus* Fieber, 1860
19	*Tropistethus fasciatus* Ferrari, 1874
Distribution	SPA: Barcelona Madrid Segovia* Tarragona Teruel
References	Bator (1957), Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).

20	*Tropistethus holosericus* (Scholz, 1846)
Distribution	AND POR SPA: Barcelona Burgos Cantabria Castellón* Cuenca Huesca Lérida Soria Teruel Zaragoza
References	Biodiversidad virtual (2013), Gessé and Goula (2006), Gessé et al. (1995), Péricart (1998a, 2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (1997, 2004).
New Records	Lérida: Coll de la Nou, Figols i Alinyà, 03/07/2000, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m; 05/06/2001, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m.
21	*Tropistethus pallipes* Reuter, 1902
Distribution	POR: Guarda SPA: Ávila Madrid Segovia
References	Costas (2004) and Péricart (1998a, 2001).
22	*Tropistethus subfasciatus* Ferrari , 1874
Synonymies	*Tropistethus albidipennis* Horváth, 1888
Distribution	SPA: Barcelona Gerona Tarragona
References	Péricart (1998a, 2001), Ribes J. and Goula (1995), Ribes J. et al. (2004) and Wagner (1960a).
Tribe	**Drymini** Stål, 1872
Genus	*Drymus* Fieber, 1860
Subgenus	*Drymus* Fieber, 1860
23	*Drymus latus latus* Douglas & Scott, 1871
Distribution	SPA: Huesca
References	Péricart (1998a, 2001).
24	*Drymus pilicornis* (Mulsant & Rey, 1852)
Distribution	POR: Beja Bragança SPA: Barcelona Cádiz Cuenca Gerona Lérida Madrid Tarragona
References	Péricart (1998a, 2001), Ribes J. (1974), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New Records	Barcelona: Canyelles, Barcelona, 11/04/2010, LM leg., MG det., 2♀, Berlese, *Hyparrhenia hirta*.
25	*Drymus pilipes* Fieber, 1878
Distribution	SPA: Barcelona Tarragona Teruel
References	Péricart (1998a, 2001) and Ribes J. and Ribes E. (2001).
26	*Drymus scambus* Stål, 1872
Distribution	SPA: Cádiz Madrid
References	Péricart (1998a, 2001) and Ribes J. (1967).
Subgenus	*Sylvadrymus* Le Quesne, 1956
27	*Drymus assimilis* Horváth, 1897
Distribution	SPA: Cádiz
References	Péricart (2001) and Ribes J. (1971).
28	*Drymus brunneus brunneus* (Sahlberg, 1848)
Distribution	AND SPA: Lérida Madrid
References	Péricart (1998a, 2001), Ribes J. (1982a, 1982b) and Ribes J. et al. (2004).
29	*Drymus brunneus confinis* Reuter, 1893
Distribution	SPA?
References	Péricart (2001).
30	*Drymus ryeii* Douglas & Scott, 1865
Distribution	SPA: Barcelona Burgos Castellón* La Coruña* León Murcia*
References	Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).

31 *Drymus scambus* Stål, 1872

Distribution SPA: Cádiz Madrid

References Péricart (1998a, 2001).

32 *Drymus sylvaticus* (Fabricius, 1775)

Distribution AND SPA: Asturias Barcelona Cantabria Gerona Huesca La Coruña Lérida Lugo Madrid Pontevedra*

References Biodiversidad virtual (2013), Gessé et al. (1994), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).

Genus *Eremocoris* Fieber, 1860

33 *Eremocoris abietis* (Linnaeus, 1758)

Distribution POR SPA: Alicante* Baleares Burgos Cáceres Castellón Huesca La Coruña León Lérida Madrid Pontevedra* Segovia Soria Tarragona* Teruel Toledo

References Biodiversidad virtual (2013), Costas et al. (1992), Péricart (1998a, 2001), Ribes J. (1972) and Ribes J. et al. (2004).

New Records Lérida: Campolado, 22/06/2010, LM leg. & det., 1♀. Coll d'Ares, Figols i Alinyà, 03/07/2000, A. Serra leg., LM det., 2♀, Pitfall, Pine forest, 1680 m.; 05/10/2000, A. Serra leg., LM det., 1♀ 1♂, Pitfall, Pine forest, 1680 m.; 05/06/2001, A. Serra leg., LM det., 5♀ 3♂, Pitfall, Pine forest, 1680 m.

34 *Eremocoris fenestratus* (Herrich-Schaeffer, 1839)

Distribution SPA: Albacete Alicante Almería* Baleares Barcelona Cáceres Cádiz Castellón Cuenca* Gerona Huesca* Madrid Murcia Pontevedra Sevilla Tarragona Teruel Valencia* Zaragoza

References Biodiversidad virtual (2013), Heteroptera from the Iberian Peninsula (2013), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes E. et al. (2000), Ribes J. (1965, 1979, 1984a, 1993), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Vázquez et al. (2003).

New Records Barcelona: Avinyó Nou , Avinyonet del Penedès, 19/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard.

35 *Eremocoris plebejus* (Fallén, 1807)

Distribution SPA: Ávila Baleares Barcelona Burgos Castellón Huesca Lérida! Madrid Segovia Teruel

References Bator (1957), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).

New Records Lérida: Devesa de la Sala, Figols i Alinyà, 04/09/2000, A. Serra leg., LM det., 1♀, Pitfall, Shrubland, 1030 m.

36 *Eremocoris podagricus* (Fabricius, 1775)

Synonymies *Rhyparochromus alpinus* Garbiglietti, 1869
Eremocoris podagricus alpinus

Distribution AND SPA: Baleares* Barcelona Castellón* Lérida! Madrid Murcia* Tarragona Valencia*

References Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. (1982b), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).

New Records Lérida: Devesa de la Sala, Figols i Alinyà, 05/06/2001, A. Serra leg., LM det., 1♀, Pitfall, Shrubland, 1030 m.

37 *Eremocoris ribauti* Vidal, 1936

Distribution SPA: Barcelona

References Ribes E. and Ribes J. (2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004, 2008).

Genus	*Gastrodes* Westwood, 1840
38	*Gastrodes abietum* Bergroth 1914
Distribution	SPA: Lérida Zaragoza
References	Péricart (1998a, 2001), Ribes J. (1982a, 1982b) and Ribes J. et al. (2004).
39	*Gastrodes grossipes grossipes* (De Geer, 1773)
Distribution	AND POR: Lisboa Porto SPA: Alicante* Ávila Barcelona Gerona Guadalajara Lérida Madrid Navarra Segovia Soria Tarragona Teruel
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Gessé (2011), Gessé and Goula (2006), Péricart (1998a, 2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New Records	Lérida: Planes de Son, Son, 25/06/2010, LM leg. & det., 1♂. Navarra: Ochagavia, 22/07/1985, A. Carapezza leg. & det.
Genus	*Ischnocoris* Fieber, 1860
40	*Ischnocoris angustulus* (Boheman, 1852)
Distribution	AND POR: Bragança Coimbra Guarda SPA: Ávila Barcelona Cádiz Ciudad Real Gerona Granada La Coruña La Rioja Lérida Lugo Madrid Pontevedra Soria Zamora
References	Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. (1974), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
41	*Ischnocoris flavipes* Signoret, 1865
Synonymies	*Ischnocoris punctulatus flavipes*
Distribution	SPA: Alicante Cáceres Castellón Ciudad Real Granada Huelva Madrid Málaga Segovia Tarragona Teruel Valencia
References	Péricart (1998a, 2001), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960b).
Observations	The specimen described in Goula & Mata (2011) has been reassigned to *Ischnocoris mundus*.
42	*Ischnocoris hemipterus* (Schilling, 1829)
Distribution	POR: Braga Bragança Guarda SPA: Barcelona Lérida Madrid
References	Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
43	*Ischnocoris mundus* (Walker, 1872)
Distribution	SPA: Barcelona Madrid
References	Goula and Mata (2011), Mata et al. (*unpublished manuscript*) and Péricart (1998a, 2001).
44	*Ischnocoris punctulatus* Fieber, 1861
Distribution	POR: Bragança SPA: Barcelona Cuenca Lérida Tarragona Teruel
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).
Genus	*Notochilus* Fieber, 1860
45	*Notochilus crassicornis* (Baerensprung, 1858)
Distribution	SPA: Alicante Ávila Baleares Barcelona Burgos Castellón Cuenca Huesca La Rioja Madrid Málaga Murcia Orense Soria Tarragona Teruel Valencia Zaragoza
References	Alonso (1983), Costas (2004), Péricart (1998a, 2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
46	*Notochilus damryi* Puton, 1871
Synonymies	*Ribautocoris humilis* *Taphropeltus humilis* Ribaut, 1929
Distribution	POR: Beja Braga Bragança Coimbra Guarda SPA: Albacete Ávila Barcelona Castellón Gerona Lérida Madrid Murcia Orense Segovia Tarragona Toledo
References	Alonso (1983), Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. (1982a, 1982b, 1984), Ribes J. and Goula (1995) and Ribes J. et al. (2004).

47 *Notochilus ferrugineus* (Mulsant & Rey, 1852)
Distribution POR: Braga Faro SPA: Ávila Barcelona Cádiz Cuenca Gerona Lérida Madrid Murcia Orense Pontevedra Tarragona Teruel Toledo Zaragoza
References Costas (2004), Costas and Vázquez (2004), Español (1964), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes E. et al. (2000), Ribes J. (1967), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (1997, 2004).
New records Barcelona: Castellet i La Gornal, 28/04/2013, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard. Lérida: Coll d'Ares, Figols i Alinyà, 01/08/2000, A. Serra leg., LM det., 1♀B, Pitfall, Pine forest, 1680 m. Coll de la Nou, Figols i Alinyà, 04/09/2000, A. Serra leg., LM det., 1♀B 1IS, Pitfall, Shrub- & grassland, 1300 m. Devesa de la Sala, Figols i Alinyà, 03/07/2000, A. Serra leg., LM det., 1♂B, Pitfall, Shrubland, 1030 m.
48 *Notochilus limbatus* Fieber, 1870
Synonymies *Taphropeltus limbatus*
Distribution AND POR SPA: Madrid
References Bator (1957), Costas et al. (1992) and Péricart (1998a, 2001).
Genus *Scolopostethus* Fieber, 1860
49 *Scolopostethus affinis* (Schilling, 1829)
Distribution POR: Faro Guarda Portalegre SPA: Barcelona Cantabria Castellón* Gerona Guipúzcoa Lérida Madrid Pontevedra* Segovia* Vizcaya! Zamora
References Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
New records Barcelona: La Fàbrega, El Moianès, 06/06/2012, LM & MG leg., LM det., 7x, River margin herbaceous vegetation. Cantabria: Ribera del río Deva, Los Llanos, 05/09/2012, LM leg. & det., 10♀/♂, Herbaceous vegetation. Vizcaya: Barrio de Artekona, Gordexola, 21/07/2012, LM leg. & det., 2♀B, River margin herbaceous vegetation.
50 *Scolopostethus cognatus* Fieber, 1878
Distribution POR SPA: Baleares Barcelona Gerona Guipúzcoa Madrid
References Bator (1957), Péricart (1998a, 2001), Ribes J. (1965), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
51 *Scolopostethus decoratus* (Hahn, 1833)
Synonymies *Scolopostethus brevis* Saunders, 1876
Distribution AND POR SPA: Alicante Baleares Barcelona Cádiz Ciudad Real* Gerona Granada* La Coruña* Lérida Pontevedra Tarragona Teruel* Zaragoza
References Biodiversidad virtual (2013), Gessé (2011), Gessé and Goula (2006), Gessé et al. (1994), Jiménez et al. (2003), Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes E. et al. (2000), Ribes J. (1965, 1974, 1988), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004), Vázquez et al. (2003) and Wagner (1960a).
New records Barcelona: Camí de Vallcàrquena, Figaró-Montmany, 13/08/2012, LM leg. & det., 2♂, Herbaceous vegetation. Carrer Roser 4, Barcelona, 05/11/2009, LM leg. & det., 1♀. Riera de Vallvidrera, La Rierada, Molins de Rei, 09/2012, A. Maceda leg., LM det., 2♀. Pontevedra: Lago Castañeiras, 14/09/2012, LM leg. & det., 1♀ 1♂, Shrub & herbaceous vegetation.
52 *Scolopostethus grandis* Horváth, 1880
Synonimies *Scolopostethus pseudograndis* Wagner, 1949
Distribution SPA: Barcelona Gerona Huesca Tarragona

References	Péricart (2001) and Ribes J. (1982a, 1982b).
Observations	This species is documented in Péricart (2001) as SP?
53	*Scolopostethus patruelis* Horváth, 1892
Distribution	AND POR SPA: Alicante Almería Baleares Barcelona Burgos Cádiz Castellón Cuenca Gerona Huesca La Rioja Lérida Pontevedra Segovia Soria Tarragona Teruel Valencia Zamora
References	Alonso (1983), Péricart (1998a, 2001), Ribes E. et al. (2000), Ribes J. (1965, 1979), Ribes J. and Ribes E. (2001), Ribes J. and Goula (1995), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960a).
54	*Scolopostethus pictus* (Schilling, 1829)
Distribution	AND POR: Bragança Guarda Porto SPA: Almería* Ávila Barcelona Cádiz Cantabria Castellón* Lérida Madrid Pontevedra* Segovia Teruel Vizcaya* Zaragoza
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a).
55	*Scolopostethus pilosus pilosus* Reuter, 1874
Distribution	POR: Bragança Coimbra SPA: Albacete Barcelona Ciudad Real Cuenca Gerona Granada Lérida Madrid Segovia Tarragona Teruel Toledo Zaragoza
References	Péricart (1998a, 2001), Ribes J. et al. (2004) and Wagner (1960b).
56	*Scolopostethus puberulus* Horváth, 1887
Distribution	SPA: La Coruña Lérida
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).
57	*Scolopostethus thomsoni* Reuter, 1874
Distribution	POR SPA: Barcelona Burgos Cantabria* Castellón* Gerona Guipúzcoa León Lugo* Málaga Navarra Tarragona Teruel* Vizcaya*
References	Alonso (1983), Biodiversidad virtual (2013), Codina (1925), Péricart (1998a, 2001) and Ribes J. and Goula (1995).
Genus	*Taphropeltus* Stål, 1872
58	*Taphropeltus andrei* (Puton, 1877)
Distribution	POR: Beja Braga Coimbra Faro* SPA: Alicante Almería* Ávila Badajoz Barcelona Cáceres Cádiz Castellón Gerona Jaén Madrid Salamanca Tarragona
References	Alonso (1983), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Gessé and Goula (2006) Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1979), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
59	*Taphropeltus contractus* (Herrich-Schaeffer, 1835)
Distribution	AND POR: Coimbra Faro SPA: Almería Ávila Barcelona Cádiz Gerona Huesca Lérida Lugo Madrid Málaga Orense Pontevedra Tarragona Teruel
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a).
60	*Taphropeltus hamulatus* (Thomson, 1870)
Distribution	SPA: Barcelona
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).
61	*Taphropeltus nervosus* (Fieber, 1861)
Distribution	POR: Coimbra Lisboa Santarém Setúbal SPA: Baleares Cádiz Granada Madrid Segovia Tarragona Valencia*
References	Biodiversidad virtual (2013), Péricart (1998a, 2001) and Ribes J. et al. (2004).

Genus	*Thaumastopus* Fieber, 1870
62	*Thaumastopus marginicollis* (Lucas, 1849)
Distribution	SPA: Barcelona Cádiz Madrid
References	Péricart (1998a, 2001), Ribes J. (1967, 1990) and Ribes J. et al. (2004).
Tribe	**Gonianotini** Stål, 1872
Genus	*Aoploscelis* Fieber, 1860
63	*Aoploscelis bivirgata* (Costa, 1835)
Distribution	POR: Bragança Coimbra Faro Guarda Leiria* Portalegre SPA: Ávila Cáceres Ciudad Real Cuenca Gerona La Coruña Madrid Salamanca Teruel
References	Bator (1957), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. (1990) and Ribes J. et al. (2004).
Genus	*Aphanus* Laporte, 1833
64	*Aphanus rolandri* (Linnaeus, 1758)
Synonimies	*Calyptonotus rolandri*
Distribution	POR: Faro* Porto SPA: Alicante Almería* Asturias* Ávila Badajoz* Baleares Barcelona Burgos Cádiz Castellón* Córdoba* Cuenca* Gerona Granada* Huelva* Huesca* La Coruña* Lérida* Madrid* Málaga* Murcia* Pontevedra* Segovia* Sevilla* Tarragona* Vizcaya* Valencia Zaragoza
References	Biodiversidad virtual (2013), Codina (1925), Costas (2004), Costas and Vázquez (2004), Grosso-Silva and Soares-Vieira (2009), Heteroptera from the Iberian Peninsula (2013), Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001), Ribes J. (1965, 1988), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Wagner (1960a).
New records	Barcelona: Avinyó Nou, Avinyonet del Penedès, 30/06/2011, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard.
Genus	*Emblethis* Fieber, 1860
65	*Emblethis angustus* Montandon, 1890
Synonimies	*Emblethis sinuatus* Wagner, 1954 (also a synonym of *Emblethis verbasci*)
Distribution	POR: Faro Setúbal SPA: Albacete Alicante Almería Ávila Baleares Cáceres Cádiz Cuenca Granada Guadalajara Huelva Jaén León Lérida! Madrid Murcia Salamanca Sevilla Tarragona Teruel Zaragoza
References	Alonso (1983), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Péricart (1998b, 2001), Ribes J. (1965, 1974), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960b).
New records	Lérida: Coll de la Nou, Figols i Alinyà, 05/06/2001, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m.
66	*Emblethis ciliatus* Horváth, 1875
Distribution	SPA: Alicante Almería Cuenca Madrid Tarragona Teruel
References	Péricart (1998b, 2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
67	*Emblethis denticollis* Horváth, 1878
Synonimies	*Emblethis pallens* Reuter, 1885
Distribution	POR: Vila Real SPA: Alicante Ávila Baleares Barcelona Cáceres Cantabria Gerona Granada Huelva Jaén La Rioja León Madrid Salamanca Segovia Tarragona Teruel Valencia Zaragoza
References	Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Jiménez et al. (2003), Péricart (1998b, 2001), Ribes J. (1965, 1982a, 1982b), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Wagner (1960a, 1960b).

68	*Emblethis duplicatus* Seidenstücker, 1963
Distribution	POR: Aveiro Bragança Coimbra Guarda Setúbal SPA: Alicante Baleares Barcelona Burgos Cáceres Cádiz Castellón Cuenca León Lugo Madrid Tarragona Zaragoza
References	Péricart (1998b, 2001), Ribes J. (1965, 1974, 1982a, 1982b), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
69	*Emblethis griseus* (Wolff, 1802)
Distribution	POR: Leiria, Setúbal SPA: Albacete Alicante Ávila Baleares Barcelona Burgos Cádiz Cantabria Ciudad Real Gerona Granada Huelva La Rioja Lérida Madrid Pontevedra Teruel Valencia Zaragoza
References	Codina (1925), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001), Ribes J. (1965, 1974), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960a).
70	*Emblethis parvus* Montandon, 1890
Distribution	POR Leiria Setúbal SPA: Alicante Baleares Cádiz Huelva Madrid Pontevedra Valencia
References	Docavo et al. (1987), Péricart (1998b, 2001), Ribes J. (1965, 1979, 1984a), Ribes J. and Sauleda (1979), Vázquez et al. (2003) and Wagner (1960a).
71	*Emblethis proximus* Seidenstücker, 1967
Distribution	SPA: Baleares Burgos Cuenca Soria Tarragona Teruel Valladolid
References	Péricart (1998b, 2001), Ribes J. (1990) and Ribes J. et al. (2004, 2008).
72	*Emblethis verbasci* (Fabricius, 1803)
Synonimies	*Emblethis sinuatus* Wagner, 1954 (also synonym of *Emblethis angustus*) *Emblethis verbasci minor*
Distribution	AND POR: Setúbal SPA: Alicante Baleares Barcelona Burgos Cádiz Castellón Huesca La Rioja Lérida Madrid Soria Tarragona Vizcaya!
References	Bator (1957), Codina (1925), Gessé (2011), Gessé et al. (1995), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1965, 1971, 1982a, 1982b), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a).
New records	Barcelona: La Talaia, El Moianès, 08/06/2012, MG leg. & det., 2x, Quercus ilex. Lérida: Coll de la Nou, Figols i Alinyà, 03/07/2000, A. Serra leg., LM det., 3♂, Pitfall, Shrub- & grassland, 1300 m.; 01/08/2000, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m.; 05/06/2001, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m. Vizcaya: Barrio de Artekona, Gordexola, 21/07/2012, LM leg. & det., 1♂, River margin herbaceous vegetation.
Genus	*Gonianotus* Fieber, 1860
73	*Gonianotus galactodermus* Fieber, 1861
Distribution	SPA: Almería Castellón Ciudad Real Granada Madrid Murcia Navarra Soria Tarragona Teruel Toledo Valencia
References	Péricart (1998b, 2001), Ribes J. (1990) and Ribes J. et al. (2004).
	Granada: Orgiva, 27/07/2010, S. Reguera leg., LM det., 1♂, 300 m.
74	*Gonianotus marginepunctatus* (Wolff, 1804)
Distribution	POR: Faro SPA: Alicante Almería Ávila Baleares Barcelona Cuenca Granada Huesca Lérida Madrid Segovia Soria Teruel Zaragoza
References	Bator (1957), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001), Ribes J. (1965), Ribes J. and Sauleda (1979), Ribes J. et al. (1997) and Wagner (1960b).
New records	Lérida: Coll d'Ares, Figols i Alinyà, 01/08/2000, A. Serra leg., LM det., 1♂, Pitfall, Pine forest, 1680 m.; 07/06/2000, A. Serra leg., LM det., 1♀ 1♂, Pitfall, Shrub- & grassland, 1300 m.; 03/07/2000, A. Serra leg., LM det., 1♀ 2♂, Pitfall, Shrub- & grassland, 1300 m.; 01/08/2000, A. Serra leg., LM det., 7♂, Pitfall, Shrub- &

	grassland, 1300 m.; 04/11/2000, A. Serra leg., LM det., 2♀ 2♂, Pitfall, Shrub- & grassland, 1300 m.; 04/01/2001, A. Serra leg., LM det., 1♀, Pitfall, Shrub- & grassland, 1300 m.
Genus	*Ischnopeza* Fieber, 1860
75	*Ischnopeza hirticornis* (Herrich-Schaeffer, 1850)
Distribution	SPA: Almería Ávila Cuenca Gerona Granada Jaén Málaga Murcia
References	Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001) and Ribes J. et al. (1997, 2004).
New records	Granada: Orgiva, 29/06/2010, S. Reguera leg., LM det., 1♂B, Pitfall, 1700 m.
Genus	*Macrodema* Fieber, 1860
76	*Macrodema micropterum* (Curtis, 1836)
Distribution	AND POR: Viana do Castelo SPA: Barcelona Gerona La Coruña León Madrid Navarra! Pontevedra
References	Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
New records	Navarra: Bosque de Irati, 22/07/1985, A. Carapezza leg. & det.
Genus	*Neurocladus* Fieber, 1860
77	*Neurocladus brachiidens* (Dufour, 1851)
Distribution	POR: Guarda SPA: Alicante Burgos Córdoba* Cuenca Granada Jaén Madrid Málaga Lérida Navarra Pontevedra* Salamanca Teruel Valladolid Vizcaya Zaragoza
References	Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (1997, 2004).
Genus	*Pionosomus* Fieber, 1860
78	*Pionosomus varius* (Wolff, 1804)
Distribution	SPA: Ávila Barcelona Burgos Ciudad Real Gerona Granada Huesca Madrid Segovia Sevilla Soria Teruel
References	Costas (2004), Costas and Vázquez (2004), Péricart (1998a, 2001), Ribes J. et al. (2004) and Wagner (19060a, 1960b).
New records	Granada: Orgiva, 20/06/2010, S. Reguera leg., LM det., 1♂, Pitfall, 2200 m.
Genus	*Pterotmetus* Amyot and Serville, 1843
79	*Pterotmetus dimidiatus* Fieber, 1861
Distribution	AND POR: Aveiro Faro Guarda Viana do Castelo Viseu SPA: Ávila Barcelona Cádiz Castellón* Gerona La Coruña* Lugo Madrid Málaga Orense Pontevedra Soria Tarragona Zamora
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Jiménez et al. (2005), Péricart (1998a, 2001), Ribes E. et al. (2000), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Vázquez et al. (2003).
New records	Soria: Villaciervos, 16/07/1985, A. Carapezza leg. & det..
80	*Pterotmetus staphyliniformis* (Schilling, 1829)
Distribution	AND SPA: Álava! Asturias! Barcelona Lérida Navarra
	Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001) and Ribes J. et al. (2004).
References	Álava: Valdegovía, Valderejo NP, 31/05/2012, LM & E. Palma leg., LM & MG det., 2♀B, Herbaceous vegetation. Asturias: Valle del Lago, Somiedo NP, 07/09/2012, LM leg., LM & MG det., 1♂ 1♂B, Herbaceous vegetation. Navarra: Bosque de Irati, 22/07/1985, A. Carapezza leg. & det..
Genus	*Trapezonotus* Fieber, 1860
Subgenus	*Trapezonotus* Fieber, 1860

81	*Trapezonotus arenarius arenarius* (Linnaeus, 1758)
Distribution	AND POR: Aveiro Faro SPA: Barcelona La Rioja Lérida Granada Huesca Lérida! Madrid Salamanca Segovia Soria Teruel
References	Biodiversidad virtual (2013), Gessé et al. (1995), Péricart (1998b, 2001), Ribes J. (1972), Ribes J. and Goula (1995), Ribes J. et al. (2004) and Wagner (1960a, 1960b).
New records	Lérida: Coll de la Nou, Figols i Alinyà, 02/05/2001, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m.
82	*Trapezonotus desertus* Seidenstücker, 1951
Distribution	AND SPA: Barcelona Gerona Huesca Lérida Pontevedra* Soria Tarragona
References	Biodiversidad virtual (2013), Péricart (1998b, 2001), Ribes J. (1982a, 1982b), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
New records	Andorra: Port d'Envalira, 12/07/1985, A. Carapezza leg. & det., 2400 m.
83	*Trapezonotus dispar* Stål, 1872
Distribution	POR: Aveiro SPA: Ávila Barcelona Cantabria* Gerona Jaén Madrid Navarra Segovia Teruel
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001) and Ribes J. et al. (2004).
84	*Trapezonotus montanus* Wagner, 1957
Distribution	SPA: Ávila Granada La Rioja León
References	Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001) and Wagner (1960b).
85	*Trapezonotus ullrichi* (Fieber, 1837)
Distribution	AND POR: Aveiro Coimbra Faro Viana do Castelo SPA: Asturias* Barcelona Cantabria Córdoba Gerona Huelva* Huesca La Coruña* La Rioja Navarra Pontevedra* Salamanca Vizcaya*
References	Biodiversidad virtual (2013), Gessé et al. (1994), Péricart (1998b, 2001) and Ribes J. et al. (2004).
New records	Navarra: Bosque de Irati, 22/07/1985, A. Carapezza leg. & det.. Burguete, 21/07/1985, A. Carapezza leg. & det.. Ochagavia, 22/07/1985, A. Carapezza leg. & det..
Tribe	**Lethaeini** Stål, 1872
Genus	*Camptocera* Jakovlev, 1877
86	*Camptocera glaberrima* (Walker, 1872)
Distribution	SPA: Almería* Alicante Ciudad Real Gerona Huesca* Madrid Murcia* Tarragona Valencia* Zaragoza
References	Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. (1984a), Ribes J. and Sauleda (1979) and Ribes J. et al. (1997, 2004, 2008).
Genus	*Lethaeus* Dallas, 1852
87	*Lethaeus fulvovarius* Puton, 1884
Distribution	SPA: Almería Murcia
References	Rieger and Pagola-Carte (2008).
Tribe	**Megalonotini** Slater, 1957
Genus	*Hispanocoris* Costas and Vázquez, 1999
88	*Hispanocoris pericarti* Costas & Vázquez, 1999
Distribution	SPA: Ávila Madrid Teruel
References	Costas (2004), Costas and Vázquez (1999), and Péricart (1998b, 2001).
Genus	*Icus* Fieber, 1860

89	*Icus angularis* Fieber, 1861
Distribution	POR: Guarda SPA: Ávila Burgos Cuenca Huesca Lérida Madrid Teruel
References	Costas et al. (1992), Péricart (1998b, 2001), Ribes E. and Ribes J. (2000) and Ribes J. et al. (2004).
Genus	*Lamprodema* Fieber, 1860
90	*Lamprodema maura* (Fabricius, 1803)
Synonimies	*Lamprodema weyersi* Puton, 1887
Distribution	POR: Beja Faro* SPA: Alicante Almería* Baleares Barcelona Cádiz Ciudad Real Córdoba* Gerona Huelva Madrid Murcia Tarragona Teruel Toledo Segovia* Valencia Zamora Zaragoza
References	Biodiversidad virtual (2013), Dusmet (1897), Español (1964, 1965), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1965, 1967, 1993), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Wagner (1960a).
Genus	*Lasiocoris* Fieber, 1860
91	*Lasiocoris anomalus* (Kolenati, 1845)
Distribution	AND POR: Bragança Santarém SPA: Albacete Alicante* Almería* Baleares Barcelona Castellón Ciudad Real Granada Guadalajara Huesca** Lérida Madrid Murcia Navarra Orense* Pontevedra* Segovia Tarragona Teruel Valencia* Zaragoza
References	Biodiversidad virtual (2013), Heteroptera from the Iberian Peninsula (2013), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1965, 1967), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Wagner (1960a).
New records	Barcelona: Avinyó Nou, Avinyonet del Penedès, 12/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard. Castellet i La Gornal, 26/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard. Tarragona: Bosc de Poblet, Vimbodí I Poblet, 22/06/2012, LM leg. & det., 1♀, Herbaceous vegetation.
92	*Lasiocoris crassicornis* (Lucas,1849)
Synonimies	*Lasiocoris antennatus* Montandon, 1889
Distribution	AND SPA: Albacete Baleares? Barcelona Cáceres Tarragona
References	Gessé et al. (1994), Péricart (1998b, 2001) and Ribes J. et al. (2004).
Genus	*Megalonotus* Fieber, 1860
93	*Megalonotus antennatus* (Schilling, 1829)
Distribution	AND SPA: Lérida Vizcaya*
References	Biodiversidad virtual (2013), Péricart (1998b, 2001).
94	*Megalonotus chiragra* (Fabricius, 1794)
Distribution	AND POR SPA: Barcelona Cáceres Gerona Huesca Zaragoza
References	Costas et al. (1992), Gessé et al. (1994, 1995), Péricart (1998b, 2001), Ribes E. and Ribes J. (2000), Ribes E. et al. (2000), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (1997, 2004) and Wagner (1960a).
95	*Megalonotus dilatatus* (Herrich-Schaeffer, 1840)
Distribution	SPA: Asturias Barcelona Gerona Lérida Tarragona
References	Alonso (1983), Péricart (1998b, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
Observations	The record for Barcelona in Ribes E. et al. (2000) was transferred by Ribes J. et al. (2004) to *Megalonotus mixtus*.
96	*Megalonotus emarginatus* (Rey, 1888)
Distribution	AND SPA: Asturias Badajoz Barcelona Córdoba Gerona Madrid Navarra Pontevedra* Soria Tarragona Zaragoza*

References Biodiversidad virtual (2013), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1990), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).

New records Barcelona: Avinyó Nou, Avinyonet del Penedès, 28/04/2011, J. Torrentó leg., LM det., 3♀, Flight intercept, Vineyard; 05/05/2011, J. Torrentó leg., LM det., 4♀ 7♂, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♀ 4♂, Flight intercept, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 4♀ 2♂, Flight intercept, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 3♀, Flight intercept, Vineyard; 16/06/2011, J. Torrentó leg., LM det., 1♂, Pitfall, Vineyard. Castellet i La Gornal, 26/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 09/06/2011, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard; 30/06/2011, J. Torrentó leg., LM det., 1♂, Flight intercept, Vineyard.

97 *Megalonotus mixtus* (Horváth, 1887)

Distribution POR SPA: Alicante Barcelona Cádiz Castellón Cuenca Gerona Granada Huelva Huesca La Coruña Lérida Madrid Soria

References Péricart (1998b, 2001), Ribes E. et al. (2000), Ribes J. (1974), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960b).

98 *Megalonotus praetextatus* (Herrich-Schaeffer, 1835)

Synonimies *Megalonotus praetextatus ibericus* Wagner, 1955
Rhyparochromus praetextus

Distribution AND POR: Viana do Castelo* SPA: Álava! Almería* Asturias* Ávila Baleares Barcelona Burgos Cáceres Cádiz Gerona Granada Madrid Murcia* Pontevedra* Segovia* Tarragona Toledo* Valencia

References Alonso (1983), Bator (1957), Biodiversidad virtual (2013), Codina (1925), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Gessé et al. (1994) Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1965, 1971), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a, 1960b).

New records Álava: Valdegovía, Valderejo NP, 31/05/2012, LM & E. Palma leg., LM det., 1♂, Herbaceous vegetation. Barcelona: Avinyó Nou , Avinyonet del Penedès, 05/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Flight intercept, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 1♂, Flight intercept, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 2♀, Flight intercept, Vineyard; 16/06/2011, J. Torrentó leg., LM det., 2♀, Flight intercept, Vineyard. Castellet i La Gornal, 12/05/2011, J. Torrentó leg., LM det., 3♀ 7♂, Flight intercept, Vineyard.

99 *Megalonotus puncticollis* (Lucas, 1849)

Distribution SPA: Badajoz Baleares Barcelona Cáceres Cádiz Gerona Madrid

References Gessé and Goula (2006), Péricart (1998b, 2001), Ribes J. (1988) and Ribes J. et al.
100 (2004).

Synonimies *Megalonotus sabulicola* (Thomson, 1870)

Distribution *Megalonotus chiragra sabulicola*
POR: Bragança Faro SPA: Almería* Ávila Barcelona Cádiz Castellón* Madrid*

References Pontevedra* Segovia* Tarragona Teruel* Valencia
Alonso (1983), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1967), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a).

New records	Barcelona: Camps de Can Valls, Montseny NP, 13/08/2011, LM leg. & det., 1♀, Herbaceous vegetation.
101	*Megalonotus setosus* Puton, 1874
Distribution	POR: Portalegre SPA: Cádiz Córdoba
References	Péricart (1998b, 2001) and Ribes J. (1974).
102	*Megalonotus subtilissimus* Roubal, 1961
Distribution	SPA: Jaén
References	Péricart (1998b, 2001).
Genus	*Pezocoris* Jakovlev, 1875
103	*Pezocoris apicimacula* (Costa, 1853)
Distribution	SPA: Cáceres Ciudad Real Madrid
References	Péricart (1998b, 2001).
Genus	*Piezoscelis* Fieber, 1870
104	*Piezoscelis staphylinus* (Rambur, 1839)
Distribution	POR: Coimbra Faro Leiria SPA: Albacete Ávila Badajoz Barcelona Cáceres Cádiz Castellón Ciudad Real Córdoba Cuenca Granada Guadalajara Huelva* Jaén Lérida Madrid Málaga Murcia Salamanca Segovia Teruel Toledo
References	Bator (1957), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Fuente (1894), Péricart (1998b, 2001), Ribes J. (1986), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
Genus	*Proderus* Fieber, 1860
105	*Proderus suberythropus* (Costa, 1842)
Distribution	POR SPA: Albacete Alicante Baleares Barcelona Cádiz Castellón Ciudad Real Jaén La Coruña Lérida Madrid Málaga Murcia Sevilla Tarragona
References	Bator (1957), Biodiversidad virtual (2013), Costas et al. (1992), Gessé (2011), Péricart (1998b, 2001), Ribes J. (1965, 1967), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004, 2008).
Tribe	**Myodochini** Blanchard, 1845
Genus	*Ligyrocoris* Stål, 1872
106	*Ligyrocoris sylvestris* (Linnaeus, 1758)
Distribution	AND SPA: Huesca Lérida
References	Gessé et al. (1995), Péricart (1998b, 2001), Ribes J. (1982a, 1982b) and Ribes J. et al. (2004, 2008).
Genus	*Pachybrachius* Hahn, 1826
107	*Pachybrachius fracticollis* (Schilling, 1829)
Distribution	AND SPA: Barcelona
References	Péricart (1998b, 2001) and Ribes J. et al. (2004).
Genus	*Paraparomius* Harrington, 1980
108	*Paraparomius leptopoides* (Baerensprung, 1859)
Synonimies	*Paromius leptopoides*
Distribution	AND POR SPA: Barcelona Cantabria* Castellón Gerona Granada* Huesca Lérida Tarragona Valencia Zaragoza*
References	Biodiversidad virtual (2013), Gessé and Goula (2006), Péricart (1998b, 2001), Ribes J. (1981), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
Genus	*Paromius* Fieber, 1860
109	*Paromius gracilis* (Rambur, 1839)

Distribution	SPA: Alicante Baleares Barcelona Castellón!** Cuenca Gerona Granada Huesca* Lugo* Tarragona Teruel Valencia Zaragoza*
References	Alonso (1983), Biodiversidad virtual (2013), Gessé (2011), Gravestein (1978), Heteroptera from the Iberian Peninsula (2013), Mata et al. (*unpublished manuscript*), Péricart (1998b, 2001), Ribes E. et al. (2000), Ribes J. and Goula (1995), Ribes J. et al. (2004) and Wagner (1960a).
New records	Barcelona: Canyelles, Barcelona, 10/05/2010, LM leg. & det., 1♀ 1♂, Ruderal herbaceous vegetation; 11/05/2010, LM leg. & det., 3♀ 1♂, Dry grassland. Ermita de Bruguers, Gavà, 07/07/2010, LM leg. & det., 5♀ 5♂, *Hyparrhenia hirta*. Montjuïc, Barcelona, 23/05/2010, LM leg., MG det., 10IS, *Hyparrhenia hirta*; 29/07/2012, LM leg. & det., 3♀ 4♂ 1IS, Ruderal herbaceous vegetation. Castellón: Alcossebre, Alcala de Xivert, 24/06/2012, LM leg. & det., 5♀ 2♂, Herbaceous vegetation. Ermita de Santa Lucía, Alcossebre, Alcala de Xivert, 29/06/2011, LM leg. & det.,1♀ 2♂, *Hyparrhenia hirta*. Tarragona: Barranc de la Mare de Deu del Camí, Cambrils, 18/06/2012, LM leg. & det., 5♀ 3♂ 2IS, Ruderal herbaceous vegetation. La Llosa, Cambrils,17/06/2012, LM leg. & det., 10IS, Oldfield herbaceous vegetation; 19/06/2012, LM leg. & det., 10IS, *Hyparrhenia hirta*. L'Ametlla de Mar, 24/09/2009, LM leg. & det., 3♀, *Hyparrhenia hirta*.
Genus	*Remaudiereana* Hoberlandt, 1954
110	*Remaudiereana annulipes* (Baerenssprung, 1859)
Distribution	POR? SPA: Alicante Almería* Baleares Barcelona Cádiz Córdoba Granada* Huelva Murcia Sevilla* Tarragona Valencia Zaragoza*
References	Biodiversidad virtual (2013), Gravestein (1969), Péricart (1998b, 2001), Ribes J. (1988), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
Tribe	**Rhyparochromini** Amyot and Serville, 1843
Genus	*Aellopus* Wolff, 1811
111	*Aellopus atratus* (Goeze, 1778)
Distribution	AND POR: Braga Bragança Castelo Branco Coimbra Faro Porto SPA: Ávila Baleares Barcelona Cáceres* Cádiz Castellón* Ciudad Real Cuenca Gerona Granada* Guadalajara* Huesca Jaén León* Lérida Lugo* Madrid Málaga* Murcia Navarra Pontevedra Soria Teruel Valencia Zaragoza
References	Bator (1957), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001), Ribes J. (1965), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Navarra: Burguete, 21/07/1985, A. Carapezza leg. & det..
Genus	*Beosus* Amyot and Serville, 1843
112	*Beosus maritimus* (Scopoli, 1763)
Synonimies	*Beosus luscus* Fabricius, 1794
Distribution	POR: Faro* SPA: Álava* Albacete* Alicante Almería* Asturias! Ávila Badajoz* Baleares* Barcelona Burgos Cádiz* Cáceres Cádiz Cantabria Castellón Ciudad Real* Córdoba* Cuenca* Gerona Granada Guipúzcoa* Huelva* Huesca* Jaén* La Coruña* León* Lérida Madrid* Murcia* Pontevedra* Segovia* Sevilla* Soria* Tarragona Teruel* Toledo* Valencia* Valladolid* Vizcaya* Zamora* Zaragoza
References	Alonso (1983), Biodiversidad virtual (2013), Codina (1925), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Dusmet (1897), Español (1964), Gessé (2011), Gessé and Goula (2006), Heteroptera from the Iberian Peninsula (2013), Jiménez et al. (2005), Mata et al. (*unpublished manuscript*), Péricart (1998b, 2001), Piñol et al. (2008), Ribes E. and Ribes J. (2001), Ribes E. et al. (2000), Ribes J. (1971, 1981),

Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (1997, 2004) and Wagner (1960a, 1960b).

New records Asturias: Valle del Lago, Somiedo NP, 07/09/2012, LM leg. & det., 1♀, Herbaceous vegetation. Barcelona: Avinyó Nou, Avinyonet del Penedès, 28/04/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 05/05/2011, J. Torrentó leg., LM det., 3♀ 2♂, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Flight intercept, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 4♂, Flight intercept, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 6♂, Flight intercept, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 1♂, Pitfall, Vineyard; 02/06/2011, J. Torrentó leg., LM det., 1♀ 2♀, Flight intercept, Vineyard; 16/06/2011, J. Torrentó leg., LM det., 1♂, Flight intercept, Vineyard; 16/06/2011, J. Torrentó leg., LM det., 1♀ 2♂, Pitfall, Vineyard; 23/06/2011, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard; 30/06/2011, J. Torrentó leg., LM det., 2♀, Flight intercept, Vineyard; 30/06/2011, J. Torrentó leg., LM det., 1♀ 5♂, Pitfall, Vineyard. Bosc Negre, Montseny NP, 13/08/2012, LM leg. & det., 1♂, Herbaceous vegetation. Camí de Vallcàrquera, Figaró-Montmany, 13/08/2012, LM leg. & det., 1♂, Herbaceous vegetation. Can Canyameres, near Sabadell, 25/09/2010, LM leg. & det., 1♂, Ruderal herbaceous vegetation. Canyelles, Barcelona, 22/04/2010, LM leg. & det., 7♀, 7♂, Ruderal herbaceous vegetation; 02/05/2010, LM leg. & det., 1♂, Dry grassland; 11/05/2010, LM leg. & det., 1♀, 1♂, Ruderal herbaceous vegetation. Castellet i La Gornal, 28/04/2011, J. Torrentó leg., LM det., 3♀, Pitfall, Vineyard; 05/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♂, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Pitfall, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 23/06/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard; 23/06/2011, J. Torrentó leg., LM det., 1♀ 2♂, Pitfall, Vineyard; 30/06/2011, J. Torrentó leg., LM det., 1♂, Pitfall, Vineyard. Jardinet de l'Om, Les Corts, Barcelona, 07/10/2010, LM leg. & det., 1♂, Urban garden. La Fàbrega, El Moianès, 06/06/2012, MG & LM leg., MG det., 1x, Herbaceous vegetation. Mas Nualart, El Moianès, 07/06/2012, LM & MG leg., LM det., 1x, Herbaceous vegetation. Montjuïc, Barcelona, 02/05/2010, LM leg. & det., 1♀, 2♂, Ruderal herbaceous vegetation; 23/05/2010, LM leg. & det., 1♀, 1♂, Ruderal herbaceous vegetation. Puig Rodó, El Moianès, 07/06/2012, LM & MG leg., LM det., 1x, Herbaceous vegetation. Sant Llorenç de Munt i Obac NP, 3-5/2007, X. Santos leg., MG det., 3x. Santa Maria de Palautordera, 30/04/2010, LM leg., MG det., 1♀, 3♀, Field margin herbaceous vegetation. Burgos: Salas de Bureba, 21/09/2012, LM leg. & det., 1♂, Ruderal herbaceous vegetation. Cantabria: Ribera del río Deva, Los Llanos, 05/09/2012, LM leg. & det., 1♂, Herbaceous vegetation. Castellón: Mas de la Montalbana, 27/06/2012, LM leg. & det., 1♀, Herbaceous vegetation.

113 *Beosus quadripunctatus* (Müeller, 1766)

Distribution SPA: Barcelona León Tarragona

References Jiménez et al. (2003), Péricart (1998b, 2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).

Genus *Dieuches* Dohrn, 1860

114 *Dieuches armatipes* (Walker, 1872)

Distribution POR SPA: Almería* Cádiz Huelva Málaga Murcia* Sevilla*

References	Biodiversidad virtual (2013), Heteroptera from the Iberian Peninsula (2013), Péricart (1998b, 2001) and Ribes J. (1967).
Genus	*Graptopeltus* Stål, 1872
115	*Graptopeltus lynceus* (Fabricius, 1775)
Synonimies	*Raglius lynceus* *Rhyparochromus lynceus*
Distribution	AND POR: Guarda SPA: Álava! Alicante Badajoz Barcelona Gerona Granada Huelva Huesca* León Lérida Madrid Navarra Pontevedra Segovia Soria Tarragona Vizcaya* Zaragoza**
References	Bator (1957), Biodiversidad virtual (2013), Heteroptera from the Iberian Peninsula (2013), Péricart (1998b, 2001), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Álava: Valdegovía, Valderejo NP, 31/05/2012, E. Palma & LM leg., LM det., 1♀, Herbaceous vegetation.
Genus	*Microtomideus* Reuter, 1885
116	*Microtomideus carbonarius* (Rambur, 1839)
Distribution	SPA: Cádiz Granada Madrid Málaga Tarragona
References	Péricart (1998b, 2001), Ribes J. (1967) and Ribes J. et al. (2004).
117	*Microtomideus leucodermus* Fieber, 1861
Distribution	POR: Beja Coimbra SPA: Cádiz Madrid Málaga
References	Biodiversidad virtual (2013), Péricart (1998b, 2001) and Ribes J. (1967).
Genus	*Peritrechus* Fieber, 1860
118	*Peritrechus angusticollis* (Sahlberg, 1848)
Distribution	SPA: Ávila Granada
References	Costas (2004), Costas and Vázquez (2004) and Péricart (1998b, 2001).
119	*Peritrechus geniculatus* (Hahn, 1832)
Distribution	SPA: Ávila Baleares Barcelona Cantabria Gerona Granada Guadalajara Lérida Lugo Madrid Segovia Tarragona Teruel
References	Alonso (1983), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Péricart (1998b, 2001), Ribes J. (1965), Ribes J. and Goula (1995), Ribes J. et al. (2004) and Wagner (1960a).
New records	Lérida: Planes de Son, Son, 22/06/2010, LM leg. & det., 1♀. Teruel: Mosqueruela, 27/06/2012, LM leg. & det., 1♀, Herbaceous vegetation. Puerto de Linares, Linares de Mora, 29/06/2012, LM leg. & det., 2♀ 1♂, Herbaceous vegetation.
120	*Peritrechus gracilicornis* Puton, 1877
Distribution	AND POR: Coimbra Guarda Vila Real! SPA: Alicante Almería* Asturias* Baleares Barcelona Burgos Cáceres Cádiz Cantabria Ciudad Real Gerona Guadalajara* Huesca Lérida Lugo Madrid Navarra Orense* Pontevedra Salamanca Segovia* Sevilla Valencia
References	Biodiversidad virtual (2013), Español (1964), Heteroptera from the Iberian Peninsula (2013), Péricart (1998b, 2001), Ribes J. (1965, 1967, 1981), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (2004), Vázquez et al. (2003) and Wagner (1960a).
New records	Baleares: Cala Macarella, Menorca, 02/07/2012, LM leg. & det., 20 ♀/♂, Herbaceous vegetation. Cala Pregonda, Menorca, 04/07/2012, LM leg. & det., 3♀ 1♂, Herbaceous vegetation. Ets Alocs, Menorca, 05/07/2012, LM leg. & det., 10♀/♂, Herbaceous vegetation. Barcelona: Can Perepoc, Campins, Montseny NP, 21/xx/2010, LM leg., MG det., 1♂, Oldfield herbaceous vegetation. Creu Vermella,

	El Moianès, 07/06/2012, LM & MG leg., LM det., 10x, Herbaceous vegetation. Ermita de Bruguers, Gavà, 07/07/2010, LM leg. & det., 1♂, Herbaceous vegetation. La Fàbrega, El Moianès, 06/06/2012, LM & MG leg., LM det., 3x, Herbaceous vegetation. La Franquesa, El Moianès, 06/06/2012, LM & MG leg., LM det., 3x, Herbaceous vegetation. Mas Nualart, El Moianès, 07/06/2012, LM & MG leg., LM det., 18x, Herbaceous vegetation. Montbrú, El Moianès, 08/06/2012, MG leg., LM det., 2x, Grassland. Voltants Coves del Toll, El Moianès, 08/06/2012, MG leg., LM det., 1x, Herbaceous vegetation. Bragança: Montesinho, Montesinho NP, 19/09/2012, LM leg. & det., 1♀, Shrub - & grassland. Burgos: Vuelta a la Peña La Isa, Cillaperlata, 01/06/2012, E. Palma & LM leg., LM det., 5♀ 3♂, Herbaceous vegetation. Vila Real: Travassos, 18/09/2012, LM leg. & det., 1♀, Oldfield herbaceous vegetation.
121	*Peritrechus lundii* (Gmelin, 1790)
Distribution	AND POR: Beja Braga Bragança Guarda SPA: Ávila Barcelona Ciudad Real León Lérida Madrid Murcia Salamanca Segovia
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Péricart (1998b, 2001) and Ribes J. et al. (2004).
122	*Peritrechus meridionalis* Puton, 1877
Distribution	SPA: Albacete Alicante Baleares Barcelona Ciudad Real Gerona Huelva Huesca Lérida Madrid Murcia Sevilla Valencia
References	Gessé (2011), Péricart (1998b, 2001), Ribes J. (1965), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960a).
123	*Peritrechus nubilus* (Fallén, 1807)
Distribution	AND POR: Coimbra Vila Real SPA: Barcelona Cádiz Gerona Huelva Huesca Madrid Zamora
References	Gessé et al. (1995), Péricart (1998b, 2001) and Ribes J. et al. (2004).
	Andorra: El Serrat, 12/07/1985, A. Carapezza leg. & det.
Genus	*Ragliodes* Reuter, 1885
124	*Ragliodes delineatus* (Rambur, 1839)
Distribution	SPA: Almería Cádiz Granada Jaén* Málaga Murcia
References	Biodiversidad virtual (2013), Péricart (1998b, 2001) and Wagner (1960b).
New records	Granada: Orgiva, 18/05/2010, S. Reguera leg., LM det., 1♀B, Pitfall, 2200 m; 17/06/2010, S. Reguera leg., LM det., 1♀B, Pitfall, 1700 m; 27/07/2010, S. Reguera leg., LM det., 1IS, Pitfall, 300 m.
Genus	*Raglius* Stål, 1872
125	*Raglius alboacuminatus* (Goeze, 1778)
Synonimies	*Rhyparochromus alboacuminatus*
Distribution	AND POR: Santarem* Viana do Castelo* SPA: Alicante Almería* Ávila Barcelona Burgos* Cádiz Ciudad Real Gerona Granada Guadalajara* Huesca La Coruña* Lérida Madrid* Pontevedra* Segovia* Valladolid* Zamora! Zaragoza*
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Gessé et al. (1995), Péricart (1998b, 2001), Ribes J. (1974, 1982b), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960b).
New records	Barcelona: Avinyó Nou, Avinyonet del Penedès, 26/05/2011, J. Torrentó leg., LM det., 1♀, Pitfall, Vineyard. Montví de Baix, El Moianès, 07/06/2012, MG leg. & det., 1x, Herbaceous vegetation. Soler de Terrades, El Moianès, 07/06/2012, MG leg. & det., 1x, Ruderal herbaceous vegetation. Lérida: Coll de la Nou, Figols i Alinyà, 03/07/2000, A. Serra leg., LM det., 1♀, Pitfall, Shrub- & grassland, 1300 m. Zamora:

	Sant Martin de Castañeda, Lago de Sanabria NP, 20/09/2012, LM leg. & det., 1♂B, Oldfield herbaceous vegetation.
126	*Raglius confusus* (Reuter, 1886)
Synonimies	*Rhyparochromus confusus*
Distribution	SPA: Álava* Barcelona Burgos* Cantabria* Castellón* Cuenca Gerona Guadalajara* Huesca* La Rioja Lérida Madrid Navarra Salamanca* Segovia Soria* Tarragona Teruel* Zaragoza
References	Alonso (1983), Biodiversidad virtual (2013), Costas et al. (1992), Péricart (1998b, 2001), Ribes J. (1982a, 1982b), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Barcelona: Camí de Vallcàrquera, Figaró-Montmany, 13/08/2012, LM leg. & det., 1♂, Herbaceous vegetation. Can Perepoc, Campins, Montseny NP, 21/xx/2010, LM leg. & det., 1♀, Oldfield herbaceous vegetation. La Franquesa, El Moianès, 06/06/2012, LM & MG leg., LM det., 4x, Herbaceous vegetation. La Talaia, El Moianès, 08/06/2012, MG leg. & det., 1x, Herbaceous vegetation.
127	*Raglius pineti* (Herrich-Schaeffer, 1835)
Synonimies	*Aphanus pineti*
Distribution	POR: Braga Coimbra Guarda SPA: Alicante Barcelona Burgos Gerona Lérida Madrid Orense Tarragona
References	Codina (1925), Péricart (1998b, 2001), Ribes J. (1984a) and Ribes J. et al. (2004).
128	*Raglius tristis* (Fieber, 1861)
Synonimies	*Rhyparochromus inarimensis* Costa, 1862 *Rhyparochromus tristis*
Distribution	POR: Bragança SPA: Ávila Baleares Barcelona Cádiz Gerona Huelva Lérida! Madrid Tarragona*
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Gessé and Goula (2006), Mata et al. (*unpublished manuscript*), Péricart (1998b, 2001), Ribes E. et al. (2000), Ribes J. (1965, 1974), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Barcelona: Collbató, 03/07/2010, LM leg. & det., 1♂, Shrubland. Lérida: Planes de Son, Son, 22/06/2010, LM leg. & det., 1♂.
Genus	*Rhyparochromus* Hahn, 1826
129	*Rhyparochromus ibericus Baerenssprung, 1858*
Distribution	SPA: Albacete Barcelona Castellón Cuenca Granada* Jaén Madrid Soria Tarragona Teruel
References	Biodiversidad virtual (2013), Péricart (1998b, 2001), Ribes J. et al. (2004) and Wagner (1960a).
130	*Rhyparochromus phoeniceus* (Rossi, 1794)
Synonimies	*Aphanus phoeniceus* *Raglius phoeniceus*
Distribution	AND POR SPA: Asturias* Ávila Baleares Barcelona Burgos Castellón Cuenca Gerona Granada Guadalajara Huesca Lérida Madrid Navarra Segovia Soria Tarragona Teruel
References	Alonso (1983), Bator (1957), Biodiversidad virtual (2013), Codina (1925), Costas (2004), Costas and Vázquez (2004), Ribes J. (1984b), Péricart (1998b, 2001), Ribes J. (1972), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Lérida: Coll d'Ares, Figols i Alinyà, 01/08/2000, A. Serra leg., LM det., 2♀ 3♂, Pitfall, Pine forest, 1680 m. Coll de la Nou, Figols i Alinyà, 07/06/2000, A. Serra

leg., LM det., 3♀, Pitfall, Shrub- & grassland, 1300 m; 03/07/2000, A. Serra leg., LM det., 2♀, Pitfall, Shrub- & grassland, 1300 m; 01/08/2000, A. Serra leg., LM det., 10♀ 4♂, Pitfall, Shrub- & grassland, 1300 m; 04/09/2000, A. Serra leg., LM det., 1♂, Pitfall, Shrub- & grassland, 1300 m; 02/05/2001, A. Serra leg., LM det., 1♀, Pitfall, Shrub- & grassland, 1300 m; 05/06/2001, A. Serra leg., LM det., 1♀, Pitfall, Shrub- & grassland, 1300 m. L'Orquera, Aiguestortes i Estany de San Maurici NP, 20/04/2011, E. Palma leg., LM det., 1♀.

131 *Rhyparochromus pini* (Linnaeus, 1758)

Synonimies *Raglius pini*

Distribution AND POR: Bragança Castelo Branco Évora Guarda SPA: Albacete Almería* Asturias! Ávila Baleares Barcelona Castellón* Gerona Granada* Huesca León* Lérida Madrid Palencia! Pontevedra* Salamanca* Segovia* Soria Teurel* Zaragoza*

References Bator (1957), Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Costas et al. (1992), Gessé et al. (1994, 1995), Péricart (1998b, 2001), Ribes J. (1972), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a).

New records Andorra: El Serrat, 12/07/1985, A. Carapezza leg. & det. Asturias: Valle del Lago, Somiedo NP, 07/09/2012, LM leg. & det., 1♀ 1♂, Herbaceous vegetation. Gerona: Tosa de Alp, 29/06/2010, LM leg. & det., 1♀, Shrubland. Lérida: Campolado, 22/06/2012, LM leg. & det., 1♀. Planes de Son, Son, 22/06/2012, LM leg. & det., 1♀. Refugio de San Nicolas, 07/07/2009, LM leg. & det., 1♀, Bog herbaceous vegetation. Palencia: Ribera del río Pisuerga, 04/09/2012, LM leg. & det., 1♂, Herbaceous vegetation. Soria: Garray, 16/07/1985, A. Carapezza leg. & det.. Villaciervos, 16/07/1985, A. Carapezza leg. & det..

132 *Rhyparochromus sanguineus* (Douglas & Scott, 1868)

Distribution SPA: Barcelona Gerona Huesca Segovia

References Biodiversidad virtual (2013), Péricart (1998b, 2001) and Ribes J. et al. (2004).

133 *Rhyparochromus vulgaris* (Schilling, 1829)

Synonimies *Raglius vulgaris*

Distribution AND SPA: Asturias* Baleares Barcelona Burgos* Cantabria* Gerona Huesca Madrid

References Biodiversidad virtual (2013), Gessé et al. (1994), Péricart (1998b, 2001), Ribes E. et al. (2000), Ribes J. (1982b) and Ribes J. and Ribes E. (2001).

New records Barcelona: Carrer Hospital 6, Barcelona, 14/10/2008, LM leg. & det., 1♀, Building roof garden. Sant Celoni, 14/05/2008, LM leg. & det., 1♀.

Genus *Trichaphanus* Kiritschenko, 1926

134 *Trichaphanus fuentei* (Puton, 1894)

Distribution SPA: Almería Ciudad Real Granada Madrid Málaga

References Biodiversidad virtual (2013), Goula et al. (2008) and Péricart (1998b, 2001).

Genus *Xanthochilus* Stål, 1872

135 *Xanthochilus minusculus* (Reuter, 1885)

Synonimies *Pachymerus reuteri* Horváth, 1885
Rhyparochromus minusculus

Distribution POR SPA: Badajoz Baleares Barcelona Cáceres Cádiz Castellón Córdoba Cuenca Gerona Granada Lérida Madrid Pontevedra Soria Tarragona Teruel

References Alonso (1983), Bator (1957), Péricart (1998b, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1965, 1967), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a, 1960b).

New records	Gerona: Cadaques, Cap de Creus NP, 29/05/2010, LM leg. & det., 1♀, Shrubland. Tarragona: Escaladei, Serra del Montsant NP, 19/06/2012, LM leg. & det., 3♂. La Llosa, Cambrils, 17/06/2012, LM leg. & det., 1♂, Oldfield herbaceous vegetation.
136	*Xanthochilus quadratus* (Fabricius, 1798)
Synonimies	*Aphanus brevirostris* Ribaut, 1921 *Aphanus quadratus var. immaculatus* Royer, 1919 *Neoxanthohilus quadratus interruptus* Wagner, 1956 *Rhyparochromus brevirostris* *Rhyparochromus quadratus*
Distribution	AND POR: Leiria Viana do Castelo* SPA: Ávila Baleares Barcelona Burgos Cáceres Cádiz Castellón* Cuenca Gerona Granada Huelva* Huesca La Coruña* León! Lérida Lugo Madrid Orense Pontevedra Salamanca* Segovia Soria Tarragona Teruel Valladolid Vizcaya
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Gessé et al. (1995), Péricart (1998b, 2001), Ribes J. (1965), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001), Ribes J. et al. (2004) and Wagner (1960a, 1960b).
New records	Barcelona: Montví de Baix, El Moianès, 07/06/2012, MG leg. & det., 2x. Voltants Coves del Toll, El Moianès, 08/06/2012, MG leg., LM det., 1x. Burgos: Arroyo de Salas, Sierra de la Demanda, 20/07/1985, A. Carapezza leg. & det.. Pineda de la Sierra, Sierra de la Demanda, 19/07/1985, A. Carapezza leg. & det.. León: Lorenzana, 17/07/1985, A. Carapezza leg. & det.. Soria: Puerto de Oncala, 16/07/1985, A. Carapezza leg. & det.. Teruel: Mosqueruela, 27/06/2012, LM leg. & det., 3♀, Herbaceous vegetation. Puerto de Linares, Linares de Mora, 29/06/2012, LM leg. & det., 1♀, Herbaceous vegetation.
137	*Xanthochilus saturnius* (Rossi, 1790)
Synonimies	*Aphanus saturnius* *Rhyparochromus saturnius*
Distribution	POR: Bragança Lisboa* SPA: Alicante Almería Ávila Badajoz Baleares Barcelona Burgos!** Cáceres Cádiz Castellón* Ciudad Real Córdoba Cuenca Gerona Granada Guadalajara* Huelva Huesca* Madrid Málaga Murcia Segovia* Sevilla Tarragona Valencia Zaragoza*
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Gessé (2011), Gessé and Goula (2006), Gómez-Menor (1956), Heteroptera from the Iberian Peninsula (2013), Mata et al. (*unpublished manuscript*, Péricart (1998b, 2001), Ribes E. et al. (2000), Ribes J. (1965, 1967), Ribes J. and Goula (1995), Ribes J. and Sauleda (1979), Ribes J. et al. (2004) and Wagner (1960a, 1960b).
New records	Baleares: Cala Pregonda, Menorca, 04/07/2012, LM leg. & det., 1♀, Herbaceous vegetation. Barcelona: Avinyó Nou, Avinyonet del Penedès, 05/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Flight intercept, Vineyard; 12/05/2011, J. Torrentó leg., LM det., 1♀ 3♂, Flight intercept, Vineyard; 19/05/2011, J. Torrentó leg., LM det., 1♀ 1♂, Flight intercept, Vineyard; 26/05/2011, J. Torrentó leg., LM det., 1♀, Flight intercept, Vineyard. Canyelles, Barcelona, 10/05/2010, LM leg. & det., 1♂, Ruderal herbaceous vegetation. Montjuïc, Barcelona, 23/05/2010, LM leg., MG det., 1♂, Ruderal herbaceous vegetation. Burgos: Cillaperlata, 30/12/2012, LM leg. & det., 4♀ 1♂, Under the bark of a fallen pine log. Granada: Orgiva, 20/07/2010, S. Reguera leg., LM det., 1♂, Pitfall, 700 m; 15/08/2010, S. Reguera leg., LM det., 1♀, Pitfall, 400 m; 29/08/2010, S. Reguera leg., LM det., 1♀, Pitfall, 1200 m.

Tribe	**Stygnocorini** Gulde, 1937
Genus	*Acompus* Fieber, 1860
138	*Acompus laticeps* Ribaut, 1929
Distribution	SPA: Barcelona Cádiz Gerona Málaga
References	Péricart (1998a, 2001), Ribes J. (1971, 1986) and Ribes J. et al. (2004).
139	*Acompus pallipes* (Herrich-Schaeffer, 1833)
Distribution	AND POR: Coimbra Guarda SPA: Álava! Barcelona Cádiz Lérida Madrid
References	Gessé and Goula (2006), Péricart (1998a, 2001), Ribes J. (1974) and Ribes J. et al. (2004).
New records	Álava: Valdegovía, Valderejo NP, 31/05/2012, LM & E. Palma leg., LM det., 1♀B, Herbaceous vegetation.
140	*Acompus rufipes* (Wolff, 1804)
Distribution	POR: Bragança Coimbra Lisboa* SPA: Ávila Barcelona Cáceres Gerona León Segovia*
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Heteroptera from the Iberian Peninsula (2013), Péricart (1998a, 2001), Ribes J. and Goula (1995) and Ribes J. et al. (2004).
Genus	*Hyalochilus* Fieber, 1860
141	*Hyalochilus ovatulus* (Costa, 1853)
Distribution	POR: Coimbra Guarda SPA: Barcelona Cádiz Gerona Granada Valencia
References	Gessé and Goula (2006), Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1974), Ribes J. and Goula (1995), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
New records	Barcelona: Jardins de Joan Maragall, Montjuïc, Barcelona, 23/05/2010, LM leg., MG det., 5♀ 3♂, Urban park. Montjuïc, Barcelona, 29/07/2012, LM leg. & det., 1♂, Ruderal herbaceous vegetation. Palau Reial, Barcelona, 17/06/2010, LM leg. & det., 1♀, Urban park.
Genus	*Lasiosomus* Fieber, 1860
142	*Lasiosomus enervis* (Herrich-Schaeffer, 1835)
Distribution	SPA: Barcelona Huesca Lérida Murcia*
References	Biodiversidad virtual (2013), Péricart (1998a, 2001), Ribes J. (1982a, 1982b), Ribes J. and Goula (1995), Ribes J. et al. (2004) and Wagner (1960a).
Genus	*Stygnocoris* Douglas and Scott, 1865
143	*Stygnocoris faustus* Horváth, 1888
Distribution	SPA: Alicante Baleares Barcelona Castellón Gerona Granada Huelva Madrid Palencia
References	Español (1964), Péricart (1998a, 2001), Ribes J. (1965), Ribes J. and Ribes E. (2001), Ribes J. and Goula (1995), Ribes J. and Sauleda (1979) and Ribes J. et al. (2004).
144	*Stygnocoris fuligineus* (Geoffroy, 1785)
Distribution	AND POR SPA: Ávila Barcelona Cádiz Cantabria* Murcia* Segovia*
References	Biodiversidad virtual (2013), Costas (2004), Costas and Vázquez (2004), Mata et al. (*unpublished manuscript*), Péricart (1998a, 2001), Ribes E. and Ribes J. (2001), Ribes J. (1967), Ribes J. and Goula (1995), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
145	*Stygnocoris rusticus* (Fallén, 1807)
Distribution	AND POR: Guarda SPA: Gerona Huesca Lérida
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).

146	*Stygnocoris sabulosus* (Schilling, 1829)
Distribution	AND POR: Bragança Coimbra Faro SPA: Álava! Alicante* Ávila Barcelona Guipúzcoa Huesca Pontevedra
References	Biodiversidad virtual (2013), Gessé et al. (1994, 1995), Péricart (1998a, 2001), Ribes J. and Ribes E. (2001) and Ribes J. et al. (2004).
New records	Álava: Senda Altube, Gorbea NP, 26/12/2012, LM leg. & det., 3♀ 1♂, Erica sp.. Pontevedra: Lago Castañeiras, 14/09/2012, LM leg. & det., 1♀, Shrub & herbaceous vegetation.
147	*Stygnocoris similis* Wagner, 1953
Distribution	SPA: Albacete Almería Barcelona
References	Péricart (1998a, 2001) and Ribes J. et al. (2004).
148	*Stygnocoris truncatus* (Horváth, 1893)
Distribution	SPA: Cádiz
References	Péricart (1998a, 2001).
Genus	*Stygnocorisella* Hoberlandt, 1956
149	*Stygnocorisella mayeti* (Puton, 1879)
Distribution	SPA: Ciudad Real Madrid
References	Péricart (1998a, 2001).
Tribe	**Udeocorini** Sweet, 1967
Genus	*Tempyra* Stål, 1874
150	*Tempyra biguttula* Stål, 1874
Distribution	SPA: Almería* Cádiz Córdoba* Murcia*
References	Baena and Torres (2012) and Biodiversidad virtual (2013).

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Appendix II Key to the families of Heteroptera from Victoria

Mature stages

Macro- and brachypterous forms

1. Antennae shorter than head **2**
— Antennae longer than head **9**
2. Interocular region of head presenting two hyaline areolae. [Figure A2.1A] . . . . **Peloridiidae** Breddin, 1897
— Interocular region of head without hyaline areolae **3**
3. Head presenting ocelli **4**
— Head without ocelli **5**
4. Forelegs raptorial. Profemora distinctly thicker than meso- and metafemora. Labium reaching at most procoxa. [Figure A2.2] . . . . **Gelastocoridae** Kirkaldy, 1897
— Forelegs not raptorial. Profemora approx. equally thick as meso- and metafemora. Labium reaching at least metacoxa. [Figure A2.3] . . . . **Ochetridae** Kirkaldy, 1906
5. Protarsi transformed into a scoop-like natatorial structure. [Figure A2.4] . . . . **Corixidae** Leach, 1815
— Protarsi not transformed into a scoop-like natatorial structure **6**
6. Forelegs raptorial **7**
— Forelegs not raptorial **8**
7. Apex of abdomen presenting a respiratory siphon that is at least as long as the body. [Figure A2.6] . . . . **Nepidae** Latreille, 1802
— Apex of abdomen without a respiratory siphon. . . . . **Belostomatidae** Leach, 1815
One species: *Diplonychus eques* (Dufour, 1863)
8. Eyes wider than interocular region of head. Hindlegs oar-like, presenting a fringe of distinctly long hairs. Metatarsi presenting a single claw. Hemelytra presenting membrane. Body longer than 4 mm. [Figure A2.7] . . . . . **Notonectidae** Latreille, 1802
— Eyes narrower than interocular region of head. Hindlegs not oar-like, without a fringe of distinctly long hairs. Metatarsi presenting two claws. Hemelytra without membrane. Body shorter than 2 mm . . . . **Pleidae** Fieber, 1851
One species: *Paraplea halei* (Lundblad, 1933)
9. Head transversely bilobed. [Figure A2.8] . . . . **Enicocephalidae** Stål, 1860

One species: *Oncylocotis saetosus* (Jeannel, 1942)

— Head not transversely bilobed **10**

10. Juga enlarged, generally meeting in front of the clypeus [Figure A2.10] . . . . **Thaumastocoridae** Kirkaldy, 1908

— Juga not enlarged **11**

11. Pronotum and hemelytra presenting a lace-like reticulated structure. Head without ocelli. Tarsi 2-segmented. [Figure A2.12] . . . . **Tingidae** Laporte, 1832

— Pronotum and hemelytra without a lace-like reticulated structure. Head presenting or not ocelli. Tarsi 2 or 3-segmented **12**

12. Head at least three times longer than wider. Eyes placed approx. half way along the length of head. [Figure A2.13] . . . . **Hydrometridae** Billberg, 1820

— Head at most two times longer than wider. Eyes at or near the base of head **13**

13. Metatarsal claws inserted before apex of last metatarsal segment **14**

— Metatarsal claws inserted at the apex of last metatarsal segment **15**

14. Mesocoxa closer to metacoxa than to procoxa. [Figure A2.14] . . **Gerridae** Leach, 1815

One species: *Rheumatometra philarete* Kirkaldy, 1902

— Mesocoxa closer to procoxa than to metacoxa or mesocoxa equidistant between procoxa and metacoxa. . . . . **Veliidae** Amyot and Serville, 1843

15. Tarsomer I distinctly shorter than tarsomer II and pronotum and coria without punctures and hemelytra of macropterous forms without cuneus and membrane of macropterous forms without veins **16**

— Tarsomer I approx. as long or longer than tarsomer II and/or pronotum and coria presenting punctures and/or hemelytra of macropterous forms presenting cuneus and/or membranes of macropterous forms presenting veins **17**

16. Tarsi 2-segmented. Labium flanked by prominent bucculae. Body length between 1.0 and 2.0 mm. [Figure A2.11] . . . . **Hebridae** Amyot and Serville, 1843

Two species: *Hebrus axillaris* Horváth,1902 and *Merragata hackeri* Hungerford, 1934

— Tarsi 3-segmented. Labium not flanked by bucculae. Body length between 2.0 and 4.4 mm. Living on the surface of epicontinental and brackish waters. . . . . **Mesoveliidae** Douglas and Scott, 1867

One species: *Mesovelia hungerfordi* Hale, 1926

17. Rostrum curved, distinctly separated from ventrum of head and thorax **18**

— Rostrum approx. straight, not separated from ventrum of head and thorax **19**

18. Head presenting a transversal furrow between the compound eyes and ocelli. Labium 3-segmented. [Figure A2.15] . . . . **Reduviidae** Latreille, 1807

— Head without transversal furrow. Rostrum 4-segmented, the first one very short. [Figure A2.17] . . . . **Nabidae** Costa, 1853

Two species: *Nabis biformis* (Bergroth, 1927)
Nabis kinbergii Reuter, 1872

Figure A2.1 A. Peloridiidae. *Peloridium hammoniorum*. **B. Anthocoridae**. **C. Lestoniidae**. *Lestonia haustorifera*. **D. Piesmatidae**. *Mcateella elongata*. Sources: (A)Yuta Nakase (B) Malcom Storey (Discoverlife) (C) American Museum of Natural History (D) Discoverlife.

19. Head presenting ocelli **20**

— Head without ocelli **44**

20. Labium 3-segmented [Figure A2.1B] . . . . **Anthocoridae** Fieber, 1836

— Labium 4-segmented **21**

21. Anterior angles of pronotum reaching juga. [Figure A2.1C] . . . . **Lestoniidae** China and Miller, 1959

One species: *Lestonia haustorifera* China, 1955

— Anterior angles of pronotum not reaching juga **22**

22. Scutellum large and convex, generally covering completely the abdomen and hemelytra **23**

— Scutellum approx. triangular, not covering completely the abdomen and hemelytra **24**

23. In dorsal view, the head is approx. as long as the diameter of eyes. Living under the bark of *Eucalyptus camaldulensis* . . . . **Aphylidae** Bergroth, 1906

One species: *Aphylum syntheticum* Bergroth, 1906

— In dorsal view, the head is longer than the diameter of eyes. [Figure A2.16] . . . . **Scutelleridae** Leach, 1815

24. Tibia presenting numerous robust spines. . . . . **Cydnidae** Billberg, 1820

— Tibia without robust spines **25**

25. Antennae 5-segmented **26**

— Antennae 4-segmented **28**

26. Labium overreaching procoxa **27**

— Labium not reaching procoxa. [Figure A2.9] . . . . **Tessarotomidae** Stål, 1864

One species: *Lyramorpha rosea* Westwood, 1837

27 Second visible sternite presenting an anteriorly-projected elongate spine [Figure A2.18] . . . . **Acanthosomatidae** Signoret, 1863

— Sterna without projecting spines. [Figure A2.20] . . . . **Pentatomidae** Leach, 1815

28. Antenomer I and femora presenting clubbed apexes. [Figure A2.19] . . . . **Berytidae** Fieber, 1851

One species: *Chinoneides tasmaniensis* (Gross, 1950)

— Antenomer I and femora without clubbed apexes **29**

29. Body covered with a layer of whitish powdery granules . . . . **Blissidae** Stål, 1862

— Body not covered with a layer of whitish powdery granules **30**

30. Tarsi 2-segmented. Body densely covered with setigerous punctures. Scutellum distinctly shorter than claval commissure. Body length between 1.7 and 3.0 mm. [Figure A2.1D] . . . . **Piesmatidae** Amyot and Serville, 1843

— Tarsi 3-segmented. Body covered or not with setigerous punctures. Scutellum shorter, equally long or longer than claval commissure **31**

Figure A2.2 Gelastocoridae. *Nerthra sp.* Source: John Tann (Flickr)

Figure A2.3 Ochetridae. *Ochterus sp.* Source: Mark Helle (Flickr)

31. Openings of the metathoracic scent-glands presenting a bristle-like process. . . . . **Hyocephalidae** Stål, 1874

One species: *Maevius luridus* Brailovsky, 2002

— Openings of the metathoracic scent-glands without bristle-like process **32**

32. Macropterous forms: membranes presenting at least six longitudinal veins. Brachypterous forms: antenniferous tubercles inserted above the line between centroid of eyes and clypeolabral suture **33**

— Macropterous forms: membranes presenting 4-6 longitudinal veins. Brachypterous forms: antenniferous tubercles inserted on or below the line between centroid of eyes and clypeolabral suture **36**

33. Always macropterous. Membranes presenting numerous cells. Living in *Nothofagus* forest. [Figure A2.21] . . . . **Idiostolidae** Scudder, 1962

One species: *Trisecus pictus* Bergroth, 1895

— Macro- or brachypterous. Membranes of the macropterous forms without cells . . **34**

34. Openings of the metathoracic scent-glands well developed. Body not red and black aposematically colored **35**

— Openings of the metathoracic scent-glands reduced. Body red and black aposematically colored. Body length between 11.0 and 16.0 mm. Living on Sapindaceae. . . . . **Rhopalidae** Amyot & Serville, 1843

One species: *Leptocoris mitellatus* (Bergroth, 1916)

35. Pronotal posterior margin distinctly wider than diatone. Bucculae reaching at least the antenniferous tubercles. [Figure A2.23] . . . . **Coreidae** Leach, 1815

— Pronotal posterior margin approx. equally wide as diatone. Bucculae reaching at most the antenniferous tubercles. [Figure A2.22] . . . . **Alydidae** Amyot & Serville, 1843

Two species: *Melanocanthus scutellaris* (Dallas, 1852) and *Mutusca brevicornis* (Dallas, 1852)

36. Eyes reniform and/or stalked. Head approx. as long as diameter of eyes. [Figure A2.25A] . . . . **Geocoridae** Baerensprung, 1860

— Eyes convex and not stalked. Head longer than diameter of eyes **37**

37. Thorax presenting only forewings. Forewings brachypterous. Living in the litter of temperate rainforest. [Figure A2.25B] . . . . **Henicocoridae** Woodward, 1968

One species: *Henicocoris monteithi* Woodward, 1968

— Thorax presenting both fore- and hindwings. Forewings macro- or brachypterous **38**

38. Suture separating sterna IV-V curving backwards, not reaching the abdominal lateral margin. [Figure A2.26] . . . . **Rhyparochromidae** Amyot and Serville, 1843

— Suture separating sterna IV-V approx. straight, reaching the abdominal lateral margin **39**

Figure A2.4 Corixidae. *Sigara sp.* Source: Arto Muinonen (Flickr)

Figure A2.5 Cymidae. Source: Jospeh Lynn (Flickr)

39. Dorsum of body presenting punctures **40**

— Dorsum of body without punctures. [Figure A2.27] ...... **Lygaeidae** Schilling, 1829

40. Coria presenting a black transversal band across its middle region. [Figure A2.25C] .. .. **Oxycarenidae** Stål, 1862

— Coria without black transversal band across its middle region **41**

41. Pronotal lateral margins laminar .... **Artheneidae** Stål, 1872

Two species: *Dilompus robustus* Scudder, 1957 and *Dilompus woodwardi* Malipatil, 1988

— Pronotal lateral margins not laminar **42**

42. Profemora thick, presenting prominent spines. Living on grasses .... **Pachygronthidae** Stål, 1865

One species: *Stenophyella macreta* Horváth, 1914

— Profemora slender, without prominent spines **43**

43. Buccula elongate, reaching the base of head. [Figure A2.25D] .... **Cryptorhamphidae** Stål, 1859

Two species: *Cryptorhamphus orbus* Stål, 1859 and *Gonystus nasutus* Stål, 1874

— Buccula short, reaching at most the base of antenniferous tubercles. [Figure A2.5] .. .. **Cymidae** Baerensprung, 1860

44. Tarsi 2-segmented. [Figure A2.31] .... **Aradidae** Spinola, 1837

— Tarsi 3-segmented **45**

45. Body distinctly dorso-ventrally flattened. Thorax presenting only highly reduced forewings. Hematophagous non-permanent ectoparasitic species feeding on humans and other mammals. [Figure A2.28] .... **Cimicidae** Latreille, 1802

One species: *Cimex lectularius* Linnaeus, 1758

— Body not dorso-ventrally flattened. Thorax presenting fore- and hindwings **46**

46. Macro- or brachypterous. Forewings of the macropterous forms presenting cuneus. Body of brachypterous forms not red and black aposematically colored. [Figure A2.30] .... **Miridae** Hahn, 1833

— Always macropterous. Forewings without cuneus. Body red and black aposematically colored **47**

47. Middle region of the posterior region of pronotum black. [Figure A2.24] .... **Largidae** Amyot and Serville, 1843

One species: *Physopelta gutta* (Burmeister, 1834)

— Middle region of the posterior region of pronotum red. [Figure A2.29] .... **Pyrrhocoridae** Fieber, 1860

Two species: *Dindymus versicolor* (Herrich-Schaeffer, 1835)
Dysdercus sidae Montrouzier, 1861

Figure A2.6 Nepidae. *Ranatra sp.* Source: Michael Drummond

Figure A2.7 Notonectidae. *Anisops sp.* Source: Jim McLean (Flickr)

Figure A2.8 Enicocephalidae. Source: James Koh (Flickr).

Figure A2.9 Tessarotomidae. *Lyramorpha rosea* Source: Doug Anderson (Flickr).

Figure A2.10 Thaumastocoridae. *Baclozygum depressum* Source: Tony Daley (Flickr)

Figure A2.11 Hebridae. *Merragata sp.* Source: Don Loarie (Flickr)

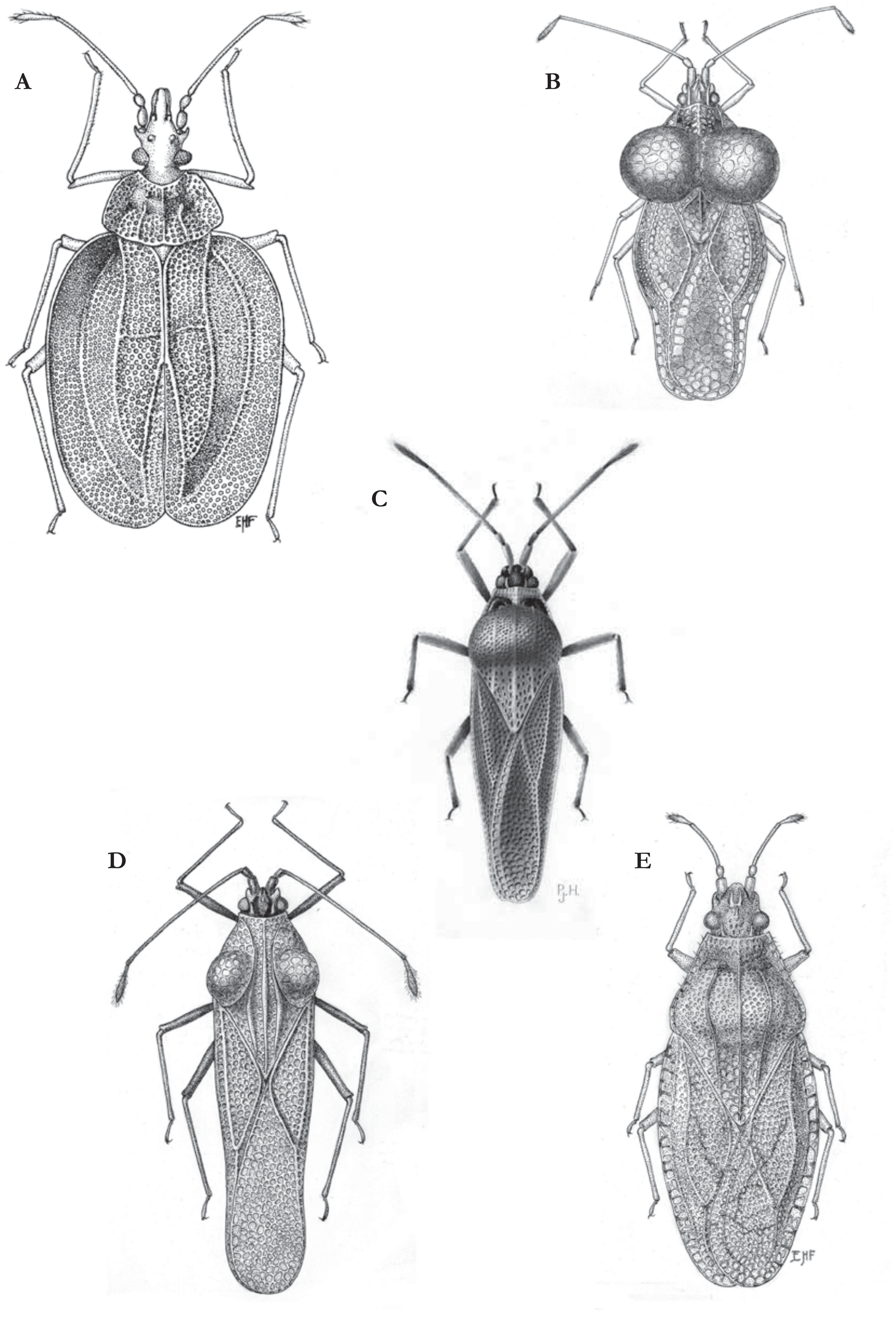

Figure A2.12 Tingidae. A. *Allocader leai.* **B.** *Diplocysta sp.* **C.** *Epimixia vittata.* **D.** *Oncophysa vesiculata.* **E.** *Euaulana ferritincta.* Sources: National Museum of Natural History and Encyclopedia of Life; (A-B-D) Elsie H. Froeschner and (C) Patricia J. Hogue.

Figure A2.13 Hydrometridae. *Hydrometra sp.* Source: Joan Quintana (Flickr)

Figure A2.14 Gerridae. Source: Jim McLean

Figure A2.15 Reduviidae. *Archilestidium cinnabarinum* Source: Nick Monaghan (Flickr).

Figure A2.16 Scutelleridae. *Choerocoris sp.* Source: original.

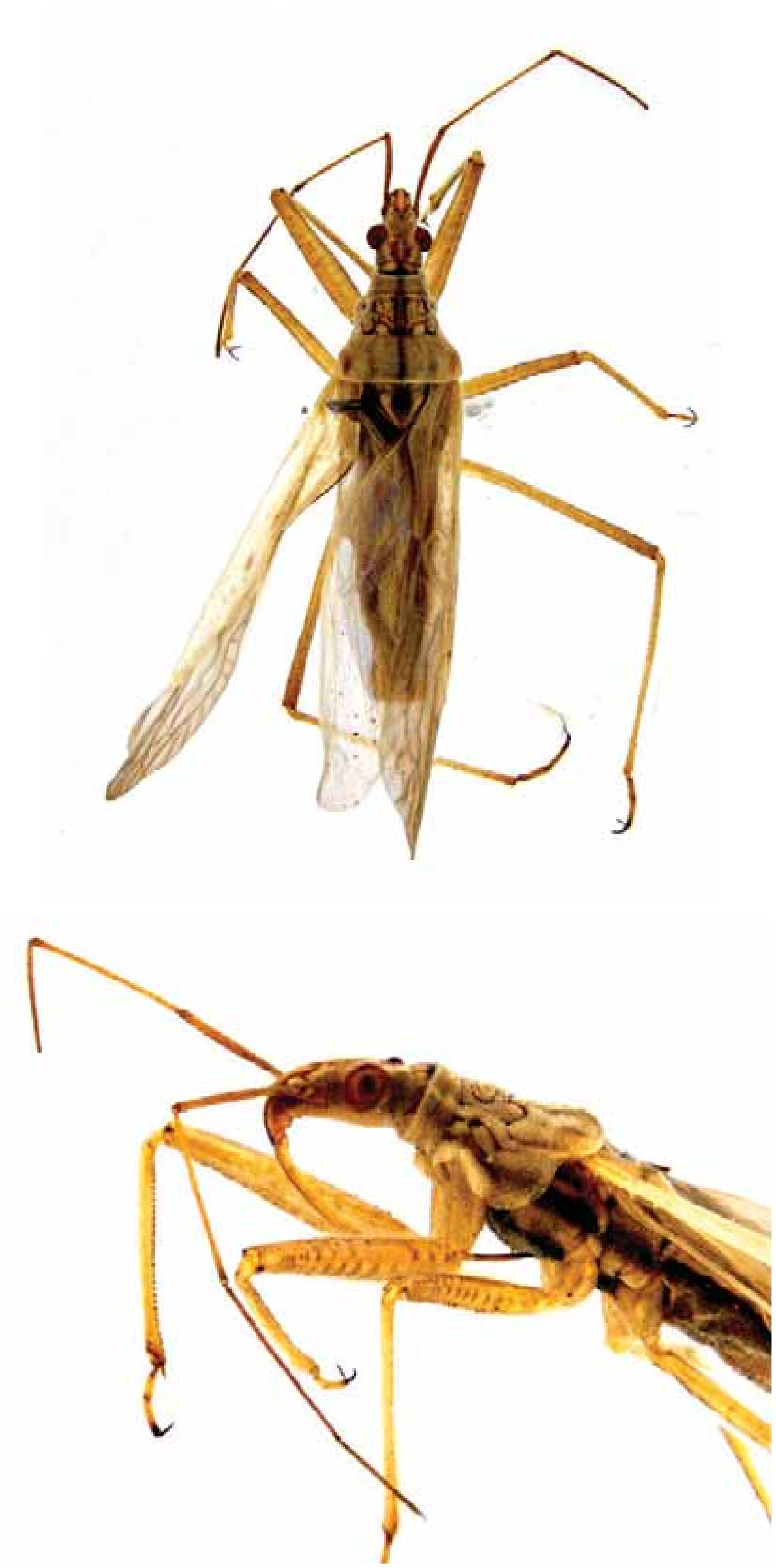

Figure A2.17 Nabidae. *Nabis kinbergii* in dorsal (top) and lateral (bottom) views. Source: Caroline Harding (Padil)

Figure A2.18 Acanthosomatidae. *Eupolemus sp.* Source: Tony Daley (Flickr).

Figure A2.19 Berytidae. *Chinoneides tasmaniensis.* Source: Tony Daley (Flickr).

Figure A2.20 Pentatomidae. *Cermatulus nasalis* (top) and *Cuspicona apothoracica* (bottom)

Source: Nick Monaghan (Flickr).

Figure A2.21 Idiostolidae. *Trisecus pictus.* Source: Kristi Ellingsen (Flickr).

Figure A2.22 Alydidae. *Mutusca brevicornis.* Source: original.

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Figure A2.23 Coreidae. *Mictis profana.* Source: Nick Monaghan (Flickr).

Figure A2.24 Largidae. *Physopelta gutta.* Source: Shipher Wu (Flickr).

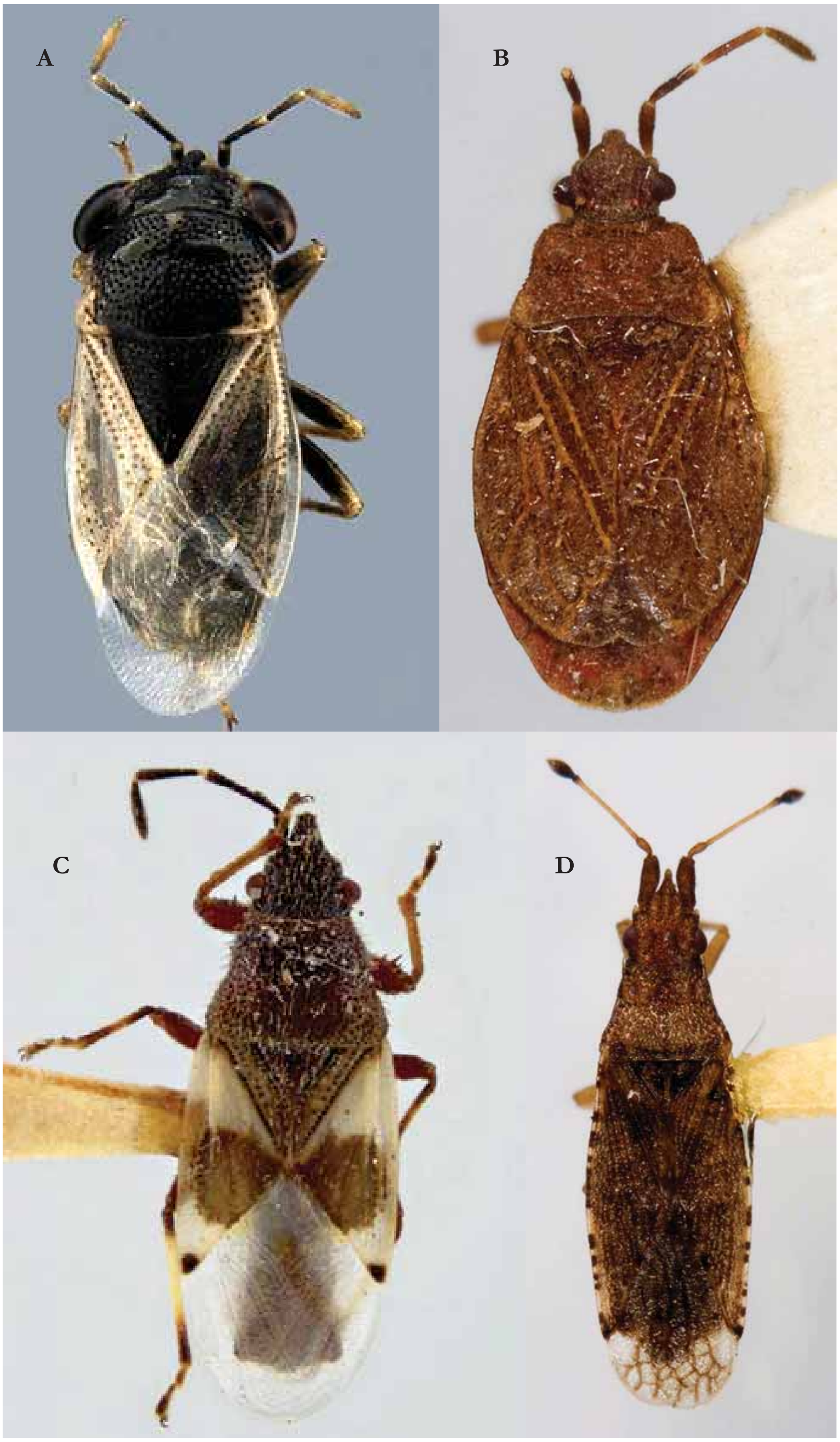

Figure A2.25 A. Geocoridae. B. Henicocoridae. *Henicocoris monteithi.* **C. Oxycarenidae.** *Oxycarenus lactuosus.* **D. Cryptorhamphidae.** *Cryptorhamphus orbus.* Sources: (A) Pia Scanlon (Padil) (B and D) American Musuem of Natural History - Discoverlife (C) Caroline Harding (Padil).

Figure A2.26 Rhyparochromidae. *Daerlac cephalotes* (top) and *Remauderiana inornata* (bottom).

Sources: Dc: Nick Monaghan (Flickr), Ri: Tony Daley (Flickr).

Figure A2.27 Lygaeidae. *Nysius vinitor* (top) and *Melanerythrus mactans* (bottom). Source: Jean and Fred Hort (Flickr).

Figure A2.28 Cimicidae. *Cimex lectularius.* Source: Gilles San Martin.

Figure A2.29 Pyrrhocoridae. *Dindymus versicolor.* Source: original

Figure A2.30 Miridae. In this page: **A.** *Coridromius monotocopsis.* **B.** *Nesidiocoris tenuis.* Opposite page: **C.** *Rayieria sp.* **D.** *Ausejanus albisignatus.* **E.** *Wallabicoris pultenaei* Sources: (A) Discoverlife (B) WonGun Kim (Flickr) (C) Mike Gordon (Flickr) (D-E) American Musuem of Natural History - Discoverlife.

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Figure A2.31 Aradidae. *Brachythynchus wilsoni* Source: Joel Bray (South Australia Musuem).

Appendix III

Key to the families of Heteroptera from the Iberian Peninsula

Mature stages

Macro- and brachypterous forms

1. Antennae longer than head **2**

— Antennae shorter than head **40**

2. Posterior margin of pronotum not expanded over scutellum **3**

— Posterior margin of pronotum expanded over scutellum, covering it partly or completely **38**

3. Antennae 4-segmented **4**

— Antennae 5-segmented **33**

4. Scutellum convex, covering almost entirely the dorsum of abdomen. Antennae, head, legs, pronotum and scutellum black. Body length approx. 2.5 mm. [Figure A3.1A] . . . . **Plataspidae** Dallas, 1851

One species: *Coptosoma scutellatum* (Geoffroy, 1785)

— Scutellum covering at most 2/3 of the dorsum of abdomen **5**

5. Head approx. 4 times longer than diatone. Living on the surface of epicontinental waters. [Figure A3.1B] . . . . **Hydrometridae** Billberg, 1820

One species: *Hydrometra stagnorum* (Linnaeus, 1758)

— Head at most 2.5 times longer than diatone **6**

6. Rostrum curved, distinctly separated from ventrum of head and thorax **7**

— Rostrum approx. straight, not separated from ventrum of head and thorax **8**

7. Head presenting a transversal furrow between the compound eyes and ocelli. Labium 3-segmented. [Figure A3.1C] . . . . **Reduviidae** Latreille, 1807

— Head without transversal furrow. Labium 4-segmented, the first one very short. [Figure A3.1D] . . . . **Nabidae** Costa, 1853

8. Metatarsi 2-segmented **9**

— Metatarsi 3-segmented **11**

9. Pronotum and hemelytra reticulated. Pronotum presenting 2, 3 or 5 longitudinal hulls. [Figure A3.2A] . . . . **Piesmatidae** Amyot and Serville, 1843

— Pronotum and hemelytra not reticulated. Pronotum without longitudinal hulls . . **10**

10. Body length between 1.1 and 2.4 mm. ♀: Brachypters. Head without ocelli. Abdomen globular. Myrmecomorphs. ♂: Macropters. Head presenting ocelli. Hemelytra presenting cuneus . . . . **Microphysidae** Dohrn, 1859

— Body length between 3.7 and 10.5 mm. ♀♂: Macropters. Body flattened. Hemelytra without cuneus. Living under the bark of trees. [Figure A3.2B] . . . . **Aradidae** Brullé, 1836

11. Labium 3-segmented **12**

— Labium 4-segmented **17**

12. Head presenting ocelli **13**

— Head without ocelli **16**

13. Ocelli separated from each other by at most the diameter of one ocelli **14**

— Ocelli separated from each other by a distance distinctly wider than the diameter of one ocelli **15**

14. Ocelli located on top of tubercles. Antenomers I-II wider than III-IV. Eyes, labium, hemelytra and profemora might present spines. [Figure A3.2C] . . . . **Leptopodidae** Brullé, 1836

— Ocelli not located on top of tubercles. Eyes might present hairs. Living on the margins of epicontinental waters. [Figure A3.2D] . . . . **Saldidae** Amyot and Serville, 1843

15. Antenomer I presenting 2 distal spines. Macropterous forms without cuneus. Body length between 2.3 and 3.5 mm. Living on the water surface. [Figure A3.3A] . . . . **Mesoveliidae** Douglas and Scott, 1867

— Antenomer I without spines. Macropterous forms presenting cuneus. Body length between 1.2 and 5.0 mm. [Figure A3.3B] . . . . **Anthocoridae** Fieber, 1836

16. Micropters. Body presenting hairs. Pronotal lateral margins laminar. Body length between 2.5 and 8.0 mm. Hematophagous endoparasitic species living on birds and mammals. [Figure A3.3C] . . . . **Cimicidae** Latreille, 1802

— Brachypters. Body without hairs. Pronotal lateral margins rounded. Body length between 2.9 and 3.6 mm. Living on the intertidal zone of the Atlantic coastline . . . . **Aepophilidae** Puton, 1879

One species: *Aepophilus bonnairei* Signoret, 1879

Notes regarding the figures:

The number between parentheses indicates the factor by which the insect has been amplified from its actual approximate size.

All illustrations by Aleksandar Stojanović.

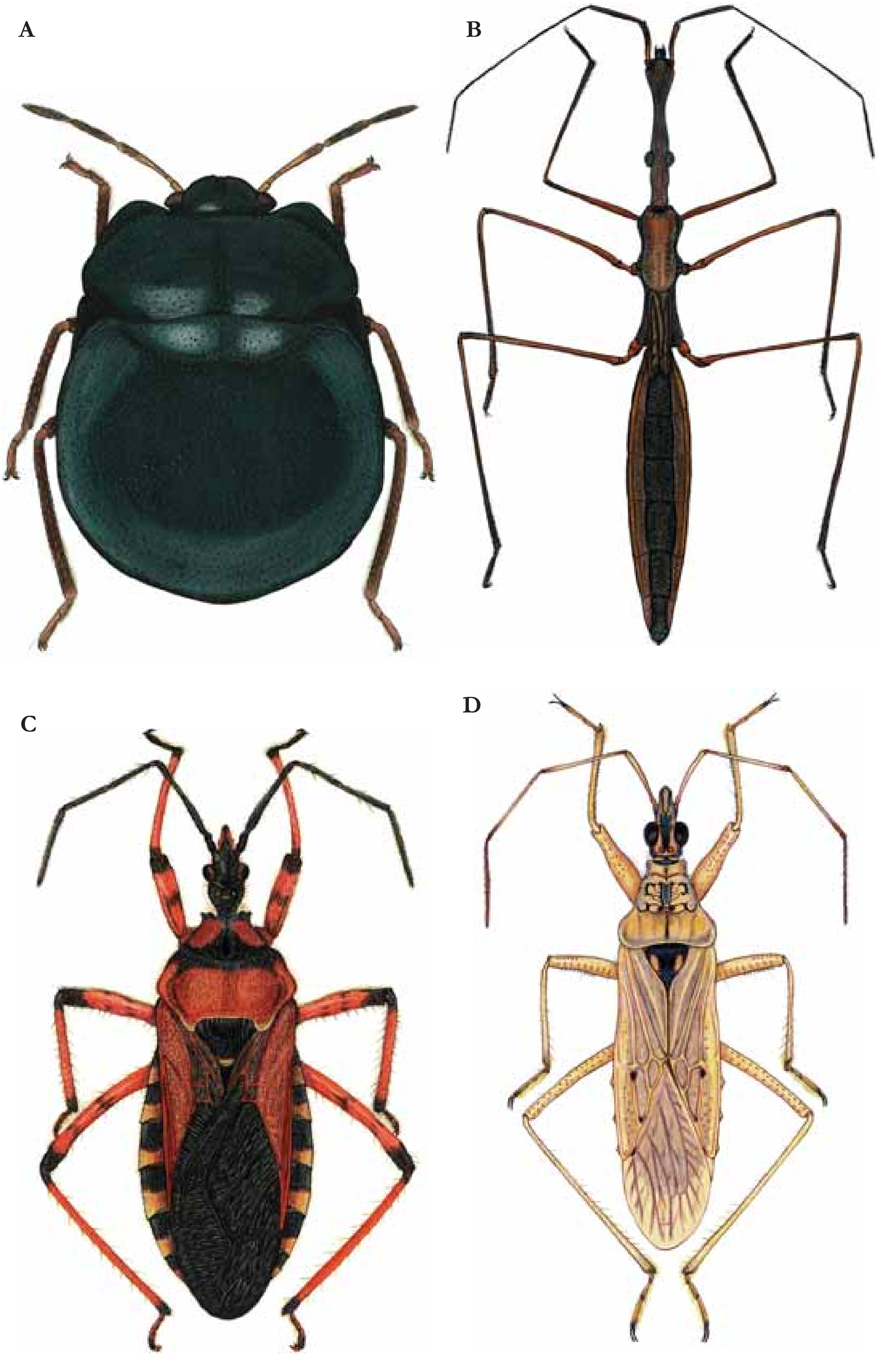

Figure A3.1 A. Plataspidae. *Coptosoma scutellatum* (x20). **B. Hydrometridae.** *Hydrometra stagnorum* (x10). **C. Reduviidae.** *Rhinocoris iracundus* (x7). **D. Nabidae.** *Nabis pseudoferus* (x15).

17. Head presenting ocelli **18**

— Head without ocelli **32**

18. Antenomers III-IV at least 2.8 times longer than antenomers I-II. Membranes of the macropterous forms without longitudinal veins. Body length between 1.1 and 2.3 mm **19**

— Antenomers III-IV shorter, approx. equal or at most 2.8 times longer than antenomers I-II. Membranes of the macropterous forms presenting longitudinal veins. Body length between 1.2 and 14.5 mm **20**

19. Labium reaching at most procoxa. Coria of the macropterous forms (♀♂) presenting towards its middle region a distinct indentation .... **Dipsocoridae** Dohrn, 1859

Two species: *Cryptostemma alienum* (Herrich-Schaeffer, 1835)
Pachycoleus pusillimum (Sahlberg, 1870)

— Labium reaching at least metacoxa. Coria of the macropterous forms (♀) without indentation. Living on moss .... **Ceratocombidae** Fieber, 1860

One species: *Ceratocombus coleoptratus* (Zetterstedt, 1819)

20. Apexes of antenomer I and femora notably dilated. Head presenting a transversal furrow between compound eyes and ocelli. [Figure A3.3D] .... **Berytidae** Fieber, 1851

— Apexes of antenomer I and/or femora not dilated. Head without a transversal furrow **21**

21. Membranes presenting at least 6 longitudinal veins **22**

— Membranes presenting at most 5 longitudinal veins **25**

Brachypterous forms .... **Annex I**

22. Antennae and metafemora presenting pale and dark rings. Juga conical and pointy, distinctly longer than clypeus. [Figure A3.4A] .... **Stenocephalidae** Latreille, 1825

— Antennae and metafemora without pale and dark rings. Juga approx. equal or shorter than clypeus **23**

23. Openings of the metathoracic scent-glands well developed **24**

— Openings of the metathoracic scent-glands reduced. Coria not distinctly sclerified between the veins. Anterior and posterior margins of the middle region of tergite V bent towards each other. [Figure A3.4B] .... **Rhopalidae** Amyot & Serville, 1843

24. Pronotal posterior margin distinctly wider than diatone. [Figure A3.4C] .... **Coreidae** Leach, 1815

— Pronotal posterior margin approx. equally wide as diatone. [Figure A3.4D] .... **Alydidae** Amyot & Serville, 1843

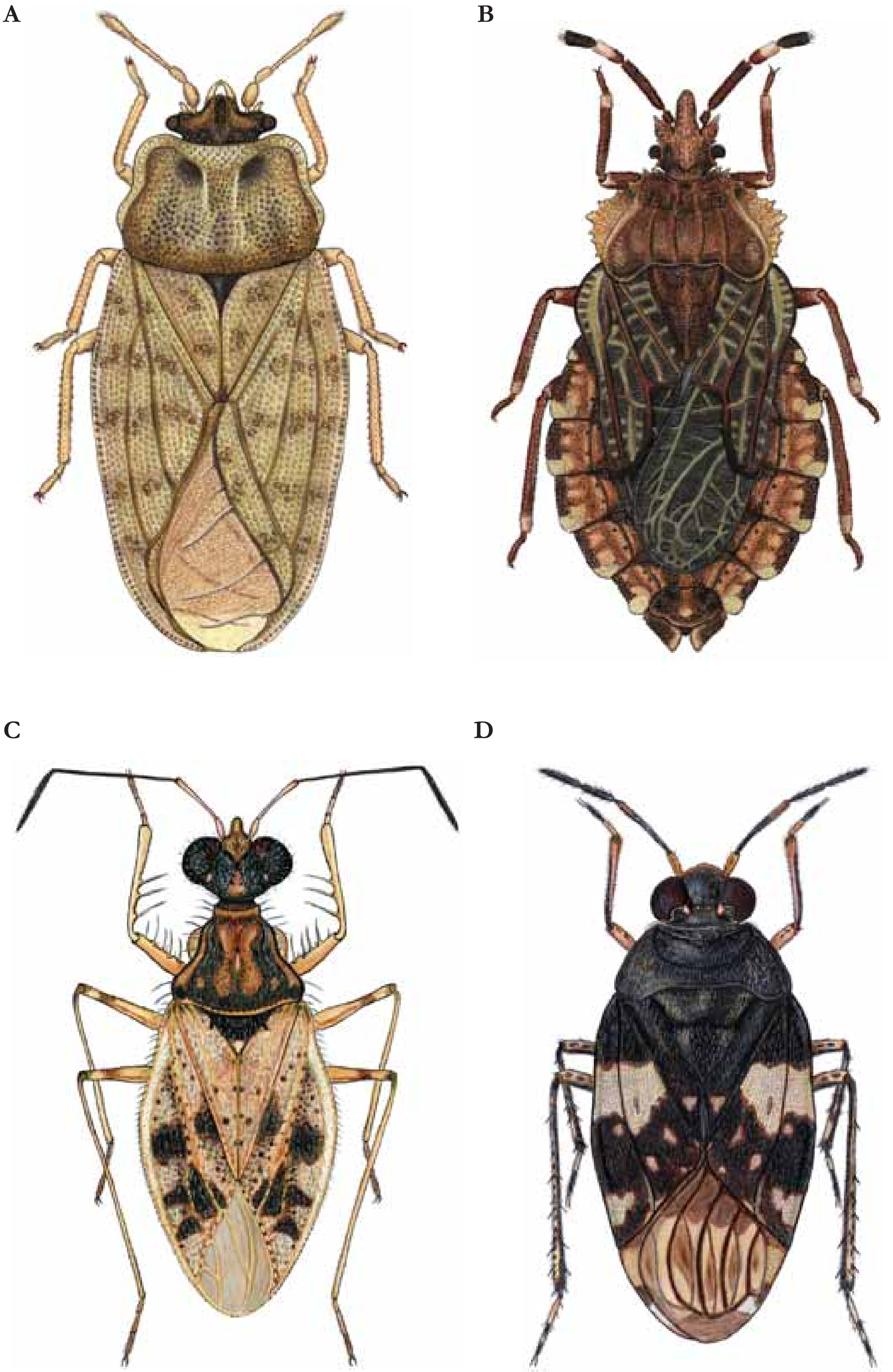

Figure A3.2 A. Piesmatidae. *Piesma capitatum* (x40). **B. Aradidae.** *Aradus betulae betulae* (x10). **C. Leptopodidae.** *Leptopus marmoratus* (x25). **D. Saldidae.** *Saldula pallipes* (x20).

25. Suture separating sterna IV-V approx. straight, reaching the abdominal lateral margin **26**

— Suture separating sterna IV-V curving backwards, not reaching the abdominal lateral margin. [Figure A3.5A] . . . . **Rhyparochromidae** Amyot and Serville, 1843

26. Anterior region of pronotum presenting a straight, arched or sinuous transversal furrow interrupted in the middle. [Figure A3.5B] . . . . **Lygaeidae** Schilling, 1829

— Anterior region of pronotum without transversal furrow **27**

27 Eyes reniform and/or stalked. Diatone wider than pronotal anterior margin. [Figure A3.6A] . . . . **Geocoridae** Baerensprung, 1860

— Eyes approx. spherical and not stalked. Diatone narrower than pronotal anterior margin **28**

28. Coria densely punctuated **29**

— Coria barely punctuated or not punctuated **31**

29. Scutellum 2-3 times longer than the commissure of clavus. [Figure A3.6B] . . . . **Heterogastridae** Stål, 1872

— Scutellum shorter or approx. equally long as the commissure of clavus **30**

30. Scutellum presenting 2 diagonal hulls, which sometimes join apically forming one single hull. . . . . **Artheneidae** Stål, 1872

— Scutellum presenting a longitudinal hull or without hulls. [Figure A3.7A] . . . . **Cymidae** Baerensprung, 1860

31. Hemelytra overreaching the abdominal lateral and posterior margins. [Figure A3.7B] . . . . **Oxycarenidae** Stål, 1862

— Hemelytra not covering the abdominal lateral and posterior margins . . . . **Blissidae** Stål, 1862

32. Coria red, presenting round black spots. Antennae, legs, head and scutellum black. Macropterous forms without cuneus. Body length between 8.0 and 9.5 mm. [Figure A3.7C] . . . . **Pyrrhocoridae** Amyot and Serville, 1843

— Coria of a wide range of colors, presenting or not round black spots. Antennae, legs, head and scutellum of a wide range of colors. Macropterous forms presenting cuneus. Body length between 1.5 and 11.8 mm. [Figure A3.7D] . . . . **Miridae** Hahn, 1833

33. Scutellum covering at most 1/6 of the length of abdomen. Body length approx. 2.0 mm. Living on the surface of epicontinental waters. [Figure A3.8A] . . . . **Hebridae** Amyot and Serville, 1843

Two species: *Hebrus depuisi* Wagner, 1954
Hebrus pusillus pusillus (Fallén, 1807)

— Scutellum covering at least 1/3 of the length of abdomen **34**

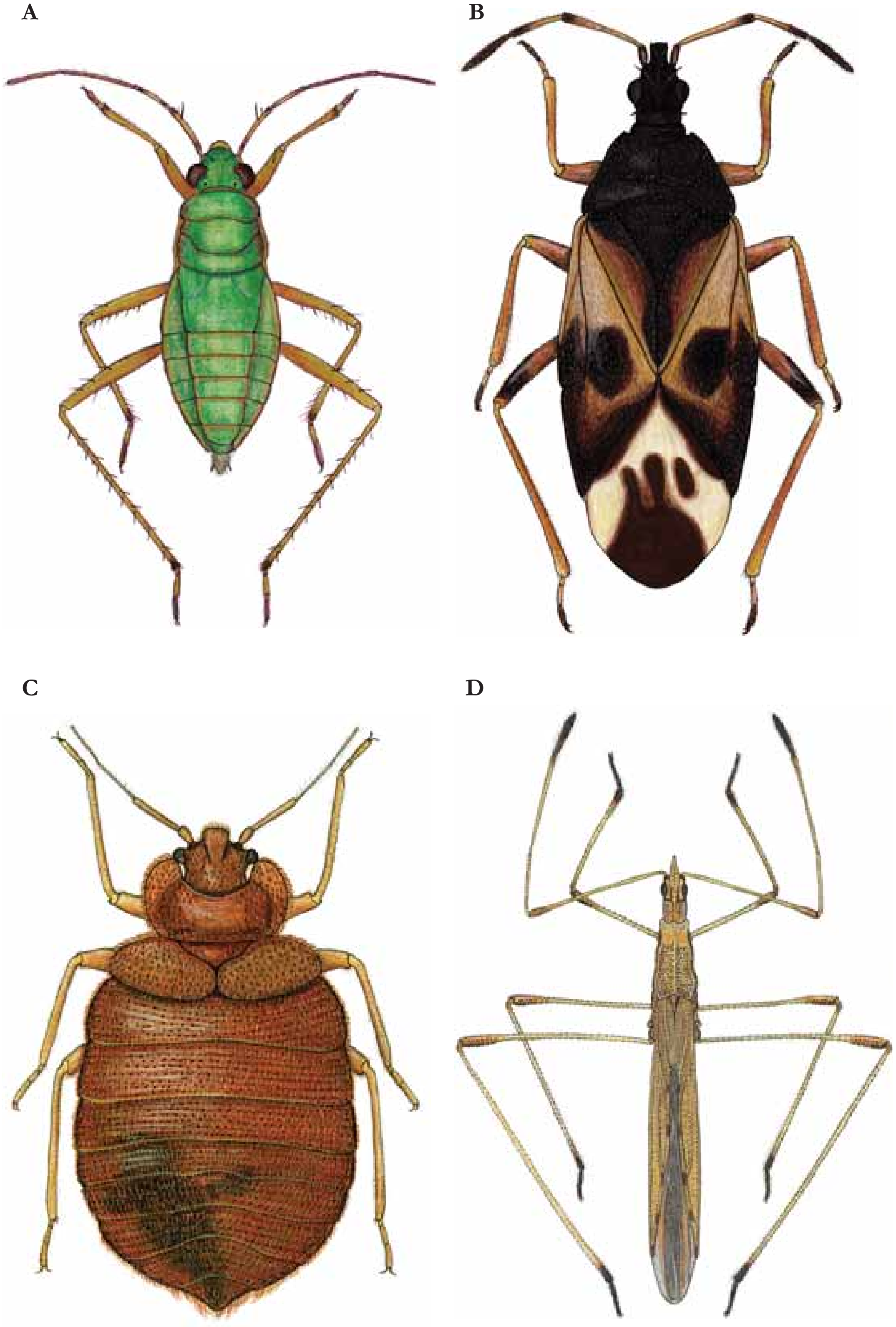

Figure A3.3 A. Mesoveliidae. *Mesovelia furcata* (x25). **B. Anthocoridae.** *Anthocoris nemorum* (x25). **C. Cimicidae.** *Cimex lectularius* (x20). **D. Berytidae.** *Neides tipularius* (x9).

34. Tarsi 2-segemented. Middle region of mesosternum presenting a distinct keel. [Figure A3.8B] . . . . **Acanthosomatidae** Signoret, 1863

— Tarsi 3-segmented. Middle region of mesosternum without a distinct keel **35**

35. Tibia presenting stout spines **36**

— Tibia without stout spines **37**

36. Scutellum oval, covering the abdomen almost entirely . . . . **Thyreocoridae** Amyot and Serville, 1843

— Scutellum triangular, covering the abdomen only partially. [Figure A3.8C] . . . . **Cydnidae** Billberg, 1820

37. Base of scutellum wider than pronotal posterior margin. Apex of scutellum always reaching to apex of abdomen. [Figure A3.8D] . . **Scutelleridae** Leach, 1815

— Base of scutellum narrower or approx. equally wide as pronotal posterior margin. Apex of scutellum reaching or not apex of abdomen. [Figure A3.9A] . . . . **Pentatomidae** Leach, 1815

38. Pronotum and hemelytra presenting a cellular and/or reticular structure. [Figure A3.9B] . . . . **Tingidae** Laporte, 1832

— Pronotum and hemelytra without a cellular and/or reticular structure **39**

39. In ventral view, mesothorax approx. 3 times longer than prothorax. Mesocoxa closer to metacoxa than procoxa. Metafemora overreaching apex of abdomen. Living on the water surface. [Figure A3.9C] . . . . **Gerridae** Leach, 1815

— In ventral view, mesothorax approx. as long as prothorax. Mesocoxa equally close to metacoxa and procoxa. Metafemora not overreaching apex of abdomen. Living on the water surface. [Figure A3.9D] . . . . **Veliidae** Brullé, 1836

40. Apex of abdomen transformed into a respiratory siphon. Living under water. [Figure A3.10A] . . . . **Nepidae** Latreille, 1802

Two species: *Nepa cinerea* Linnaeus, 1758 and
Ranatra linearis (Linnaeus, 1758)

— Apex of abdomen not transformed into a respiratory siphon **41**

41. Labium reaching at most procoxa **42**

— Labium reaching at least metacoxa **45**

42. Protarsi longer than protibia, not segmented, transformed into a swimming appendage. Labium not segmented. Living under water. [Figure A3.10B] . . . . **Corixidae** Leach, 1815

— Protarsi shorter than protibia, segmented, not transformed into a swimming appendage. Labium segmented **43**

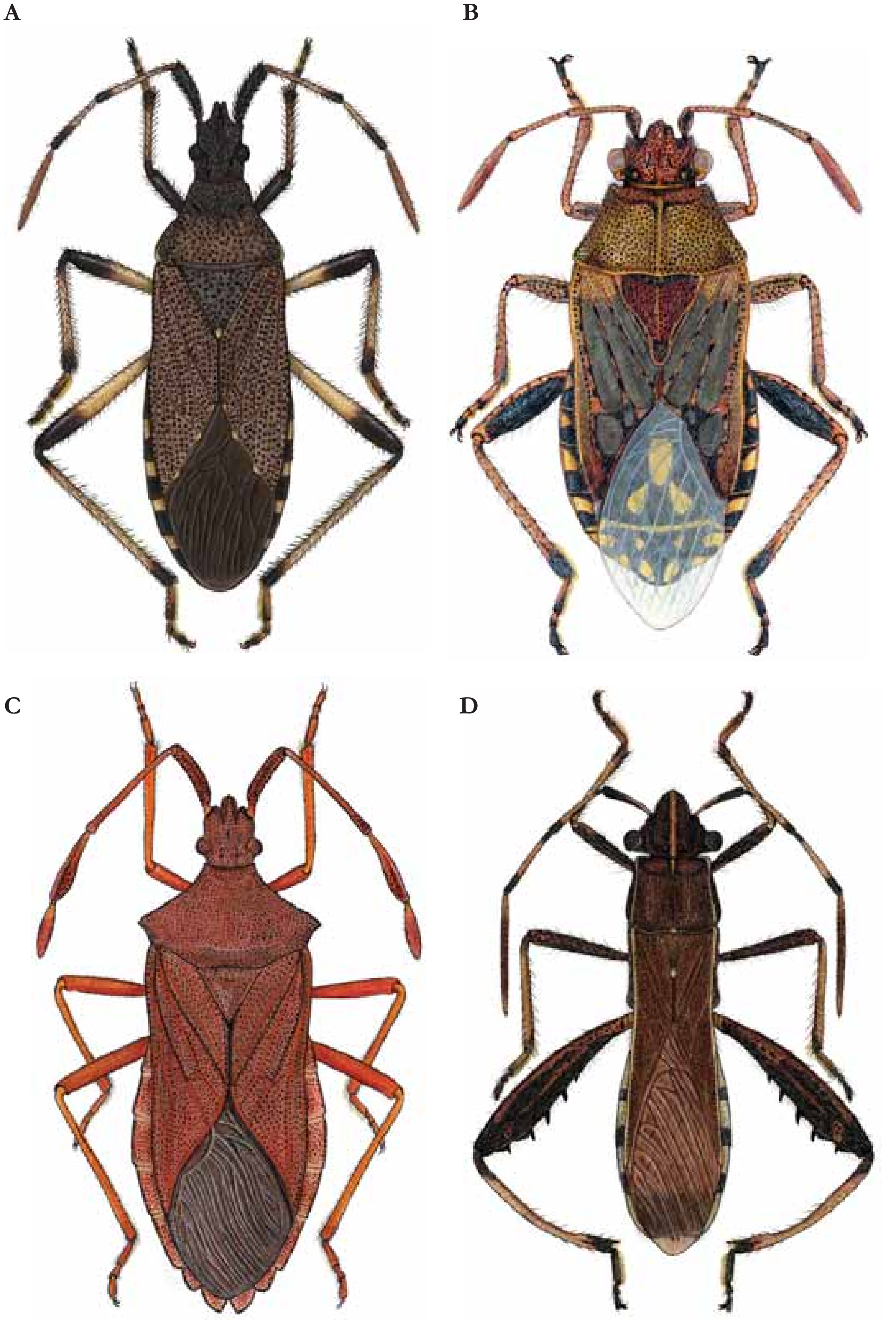

Figure A3.4 A. Stenocephalidae. *Dicranocephalus albipes* (x8). **B. Rhopalidae.** *Stictopleurus punctatonervosus* (x13). **C. Coreidae.** *Gonocerus acunteangulatus* (x8). **D. Alydidae.** *Camptopus lateralis* (x8).

43. Body flattened. Hemelytra lying flat over abdomen. Apex of profemora distinctly thickened. Protibia arched. Body length between 8.0 and 16.0 mm. Living under water. [Figure A3.10C] .... **Naucoridae** Leach, 1815

— Body compressed. Hemelytra laying roof-like over abdomen. Apex of profemora not thickened. Protibia approx. straight **44**

44. Body length between 2.0 and 3.0 mm. Living under water. [Figure A3.10D] .... **Pleidae** Fieber, 1851

One species: *Plea minutissima* Leach, 1817

Body length between 7.5 and 17.0 mm. Living under water. [Figure A3.11A] ....

— .... **Notonectidae** Latreille, 1802

Head presenting ocelli. Body length between 4.0 and 6.0 mm. Living on river and lake

45. margins. [Figure A3.11B] .... **Ochteridae** Kirkaldy, 1906

One species: *Ochterus marginatus marginatus* (Latreille, 1804)

Head without ocelli. Body length between 8.5 and 10.0 mm. Living under water ....

— .... **Aphelocheridae** Fieber, 1851

Two species: *Aphelocheirus murcius* Nieser and Millán, 1989

Aphelocheirus occidentalis Nieser and Millán, 1989

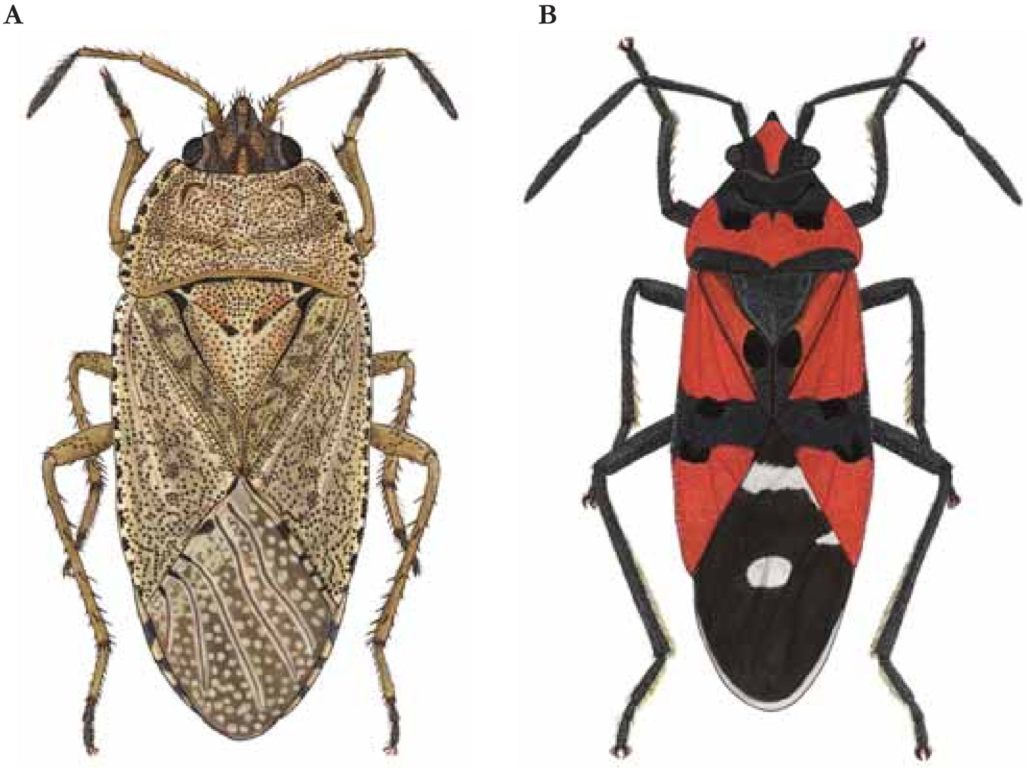

Figure A3.5 A. Rhyparochromidae. *Emblethis verbasci* (x15). **B. Lygaeidae.** *Lygaeus equestris* (x9).

Appendix III Key to the families of Heteroptera from the Iberian Peninsula

Annex 1 Brachypterous forms keying out in couplet 21 of the main key

1. Suture separating sterna IV-V curving backwards, not reaching the abdominal lateral margin . . . . **Rhyparochromidae** Amyot and Serville, 1843 (Group A)
Numerous genera: *Aoploscelis* Fieber, 1860*, Camptocera* Jakovlev, 1877*, Ischnocoris* Fieber, 1860*, Ischnopeza* Fieber, 1860*, Hispanocoris* Costas and Vázquez, 1999*, Lamprodema* Fieber, 1860*, Macrodema* Fieber, 1860*, Megalonotus* Fieber, 1860*, Notochilus* Fieber, 1860*, Piezoscelis* Fieber, 1870*, Pterotmetus* Amyot and Serville, 1843*, Raglius* Stål, 1872*, Ragliodes* Reuter, 1885*, Scolopostethus* Fieber, 1860*, Stygnocoris* Douglas and Scott, 1865*, Trapezonotus* Fieber, 1860 and *Tropistethus* Fieber, 1860

— Suture separating sterna IV-V approx. straight, reaching the abdominal lateral margin **2**

2. Coria reaching the apex of abdomen, giving the insect a coleopteroid appearance. Body length between 1.8 and 2.4 mm . . . . **Oxycarenidae** Stål, 1862
One species: *Anomaloptera helianthemi* Amyot and Serville, 1843

— Coria not reaching the apex of abdomen. Body length between 1.4 and 12.0 mm **3**

3. Posterior margin of tergite IV curved towards apex of abdomen. Anterior margin of tergite VI curved towards base of abdomen. Body length between 6.0 and 8.5 mm . . . . **Rhopalidae** Amyot & Serville, 1843
One species: *Myrmus miriformis* (Fallén, 1807)

— Posterior margin or tergite IV straight or curved towards apex of abdomen. Anterior margin of tergite VI straight or curved towards the apex of abdomen. Body length between 1.4 and 12.0 mm **4**

4. Anterior region of pronotum presenting a straight, arched or sinuous transversal furrow interrupted in the middle. Body length between 4.0 and 7.0 mm . . . . **Lygaeidae** Schilling, 1829
Four genera: *Apterola* Mulsant and Rey, 1866*, Horvathiolus* Josifov, 1965*, Lygaeosoma* Spinola, 1837 and *Nithecus* Horváth, 1890

— Anterior region of pronotum without transversal furrow. Body length between 1.4 and 12.0 mm **5**

5. Body length between 1.4 and 4.8 mm **6**

— Body length between 7.5 and 12.0 mm **8**

6. Diatone wider than pronotal posterior margin . . . . . . **Geocoridae** Baerensprung, 1860

One genus: *Geocoris* Fallén, 1814

— Diatone narrower than pronotal posterior margin **7**

7. Hemelytra yellowish-brownish to black. Coria distinctly punctuated. Body length between 1.4 and 4.5 mm . . . . **Rhyparochromidae** Amyot and Serville, 1843 (Group B)

Two genera: *Acompus* Fieber, 1860 and *Plinthisus* Stephens, 1829

— Hemelytra almost uniformly pale. Coria not punctuated or only slightly punctuated. Body length between 3.0 and 6.0 mm . . . . **Blissidae** Stål, 1862

Two genera: *Dimorphopterus* Stål, 1872 and *Ischnodemus* Fieber, 1837

8. Antenomer I approx. as thick as antenomer II. Antenomers II and III presenting pale and dark rings. Antenomer IV longer than antenomer I . . . . **Alydidae** Amyot & Serville, 1843

One species: *Mycrelytra fossularum* (Rossi, 1790)

— Antenomer I distinctly thicker than antenomer II. Antenomer II and III without pale and dark rings. Antenomer IV shorter than antenomer I . . . . . . **Coreidae** Leach, 1815

One species: *Prionotylus brevicornis* (Mulsant and Rey, 1852)

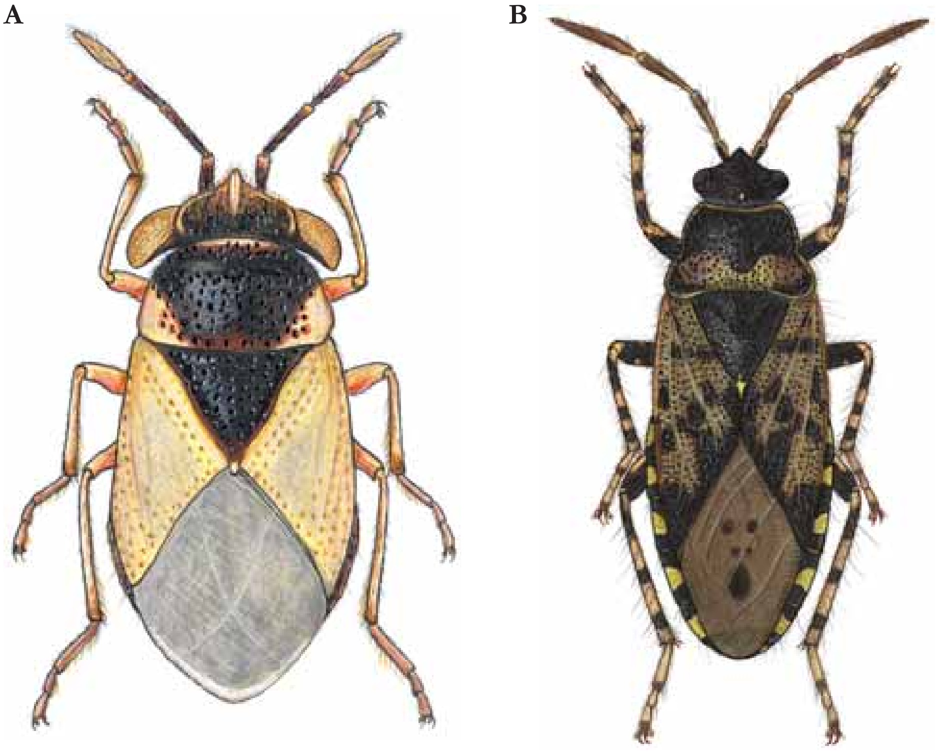

Figure A3.6 A. Geocoridae. *Geocoris megacephalus* (x25). **B. Heterogastridae.** *Heterogaster urticae* (x15).

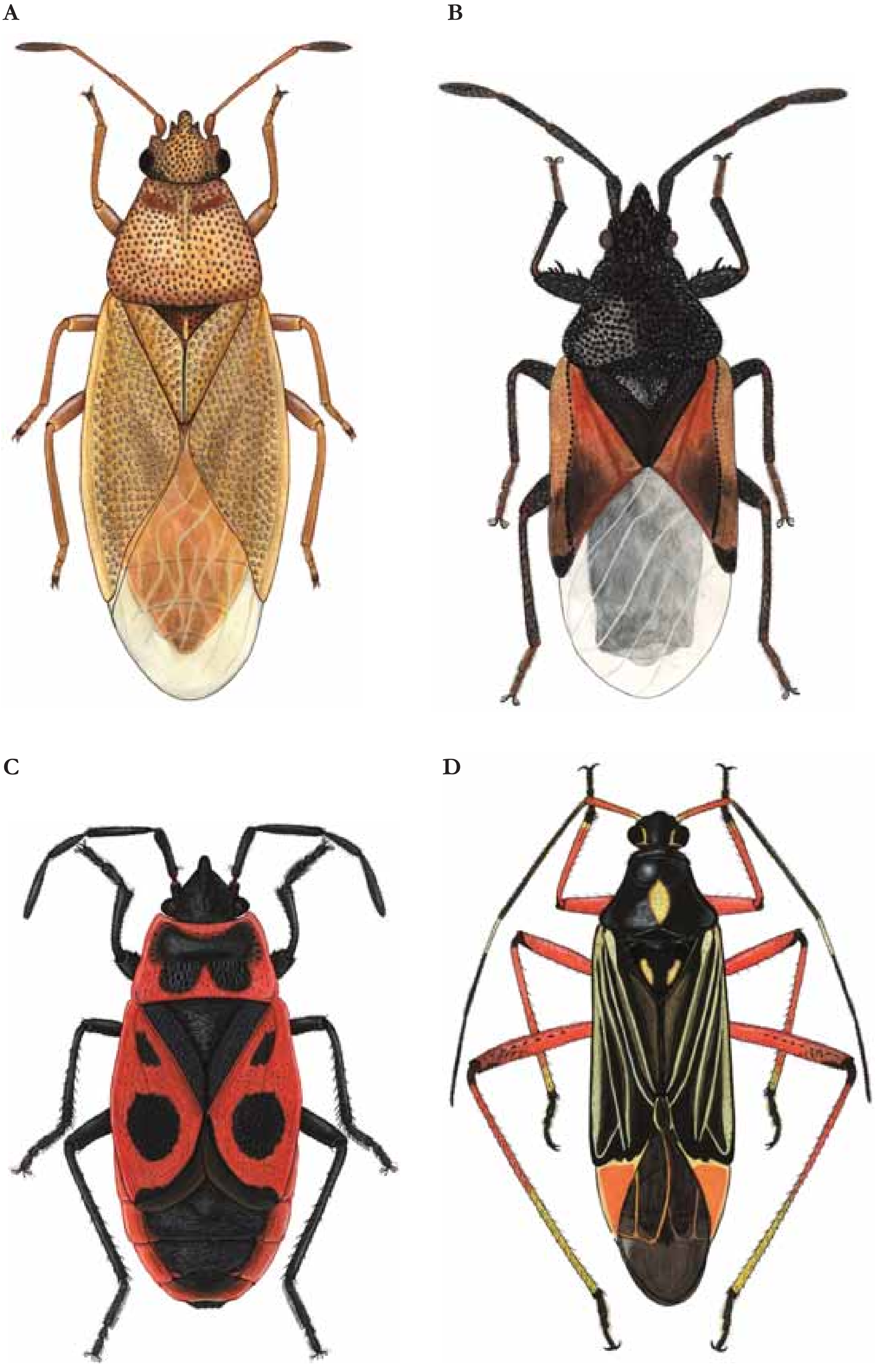

Figure A3.7 A. Cymidae. *Cymus glandicolor* (x23). **B. Oxycarenidae.** *Oxycarenus lavaterae* (x20). **C. Pyrrhocoridae.** *Pyrrhocoris apterus* (x10). **D. Miridae.** *Miris striatus* (x10).

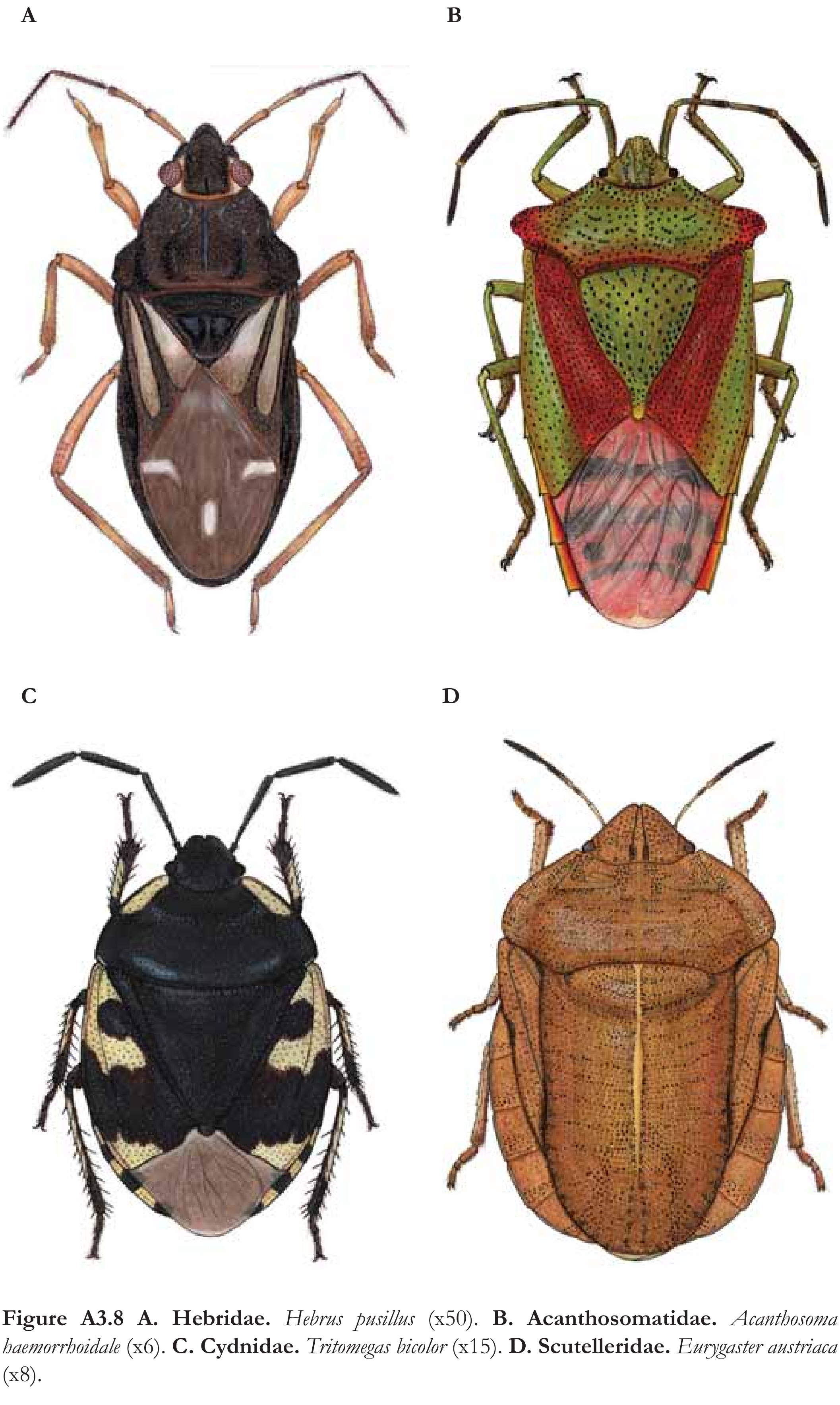

Figure A3.8 A. Hebridae. *Hebrus pusillus* (x50). **B. Acanthosomatidae.** *Acanthosoma haemorrhoidale* (x6). **C. Cydnidae.** *Tritomegas bicolor* (x15). **D. Scutelleridae.** *Eurygaster austriaca* (x8).

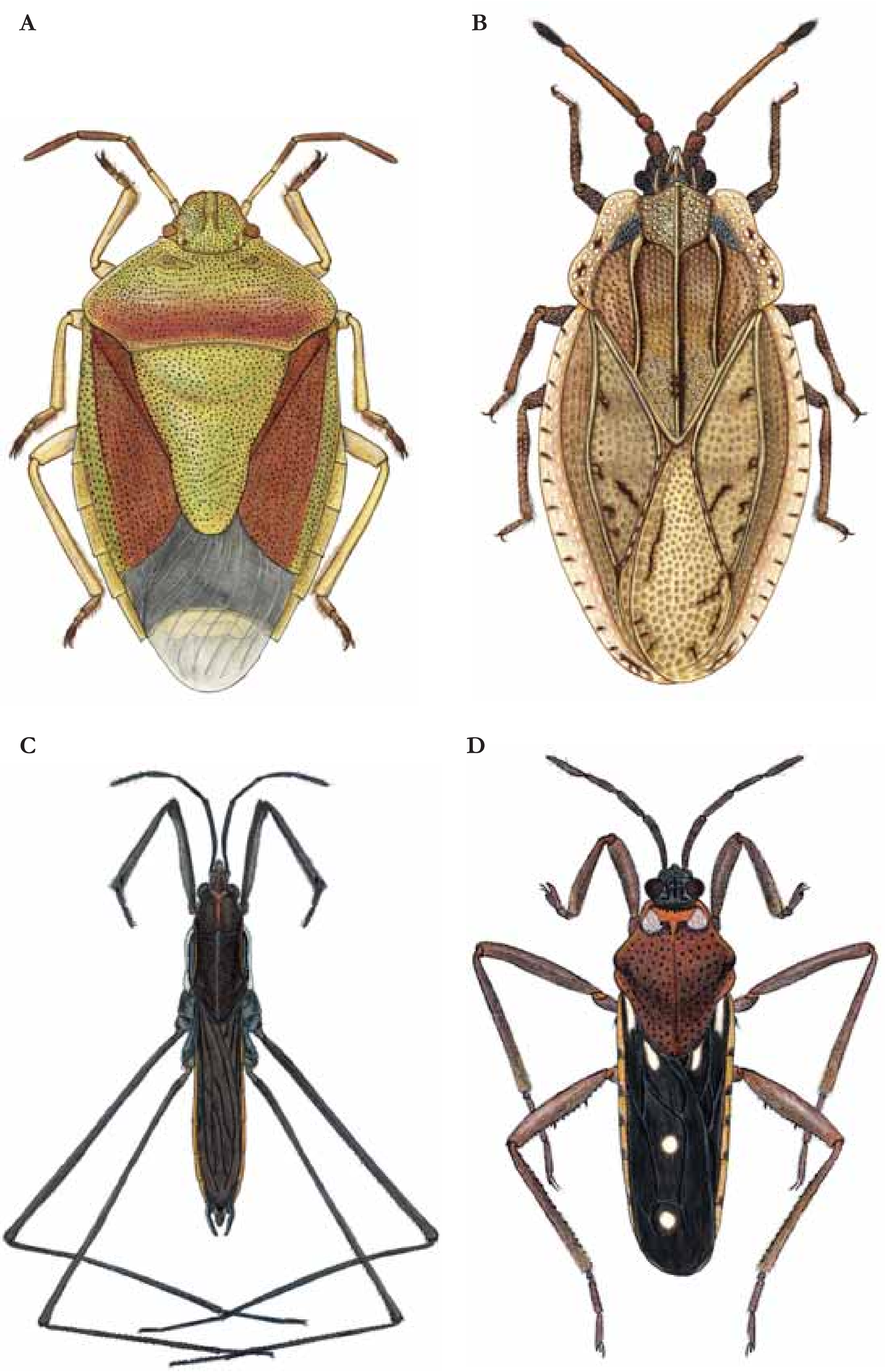

Figure A3.9 A. Pentatomidae. *Piezodorus lituratus* (x8). **B. Tingidae.** *Tingis cardui* (x33). **C. Gerridae.** *Aquarius paludum* (x7). D. **Veliidae.** *Velia currens* (x15).

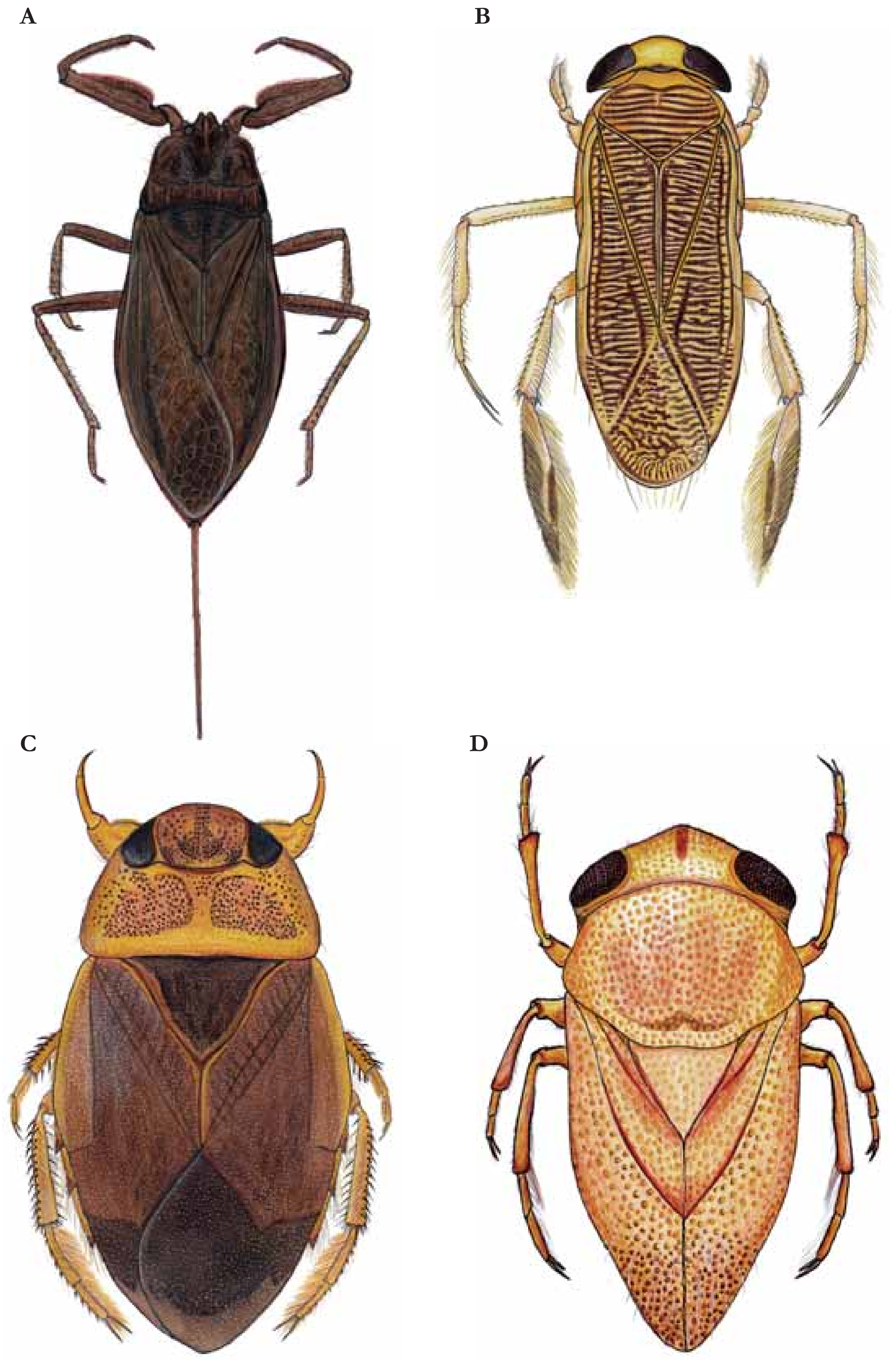

Figure A3.10 A. Nepidae. *Nepa cinerea* (x9). **B. Corixidae.** *Callicorixa praeusta* (x13) **C. Naucoridae.** *Ilyocoris cimicoides* (x9). **D. Pleidae.** *Plea minutissima* (x40).

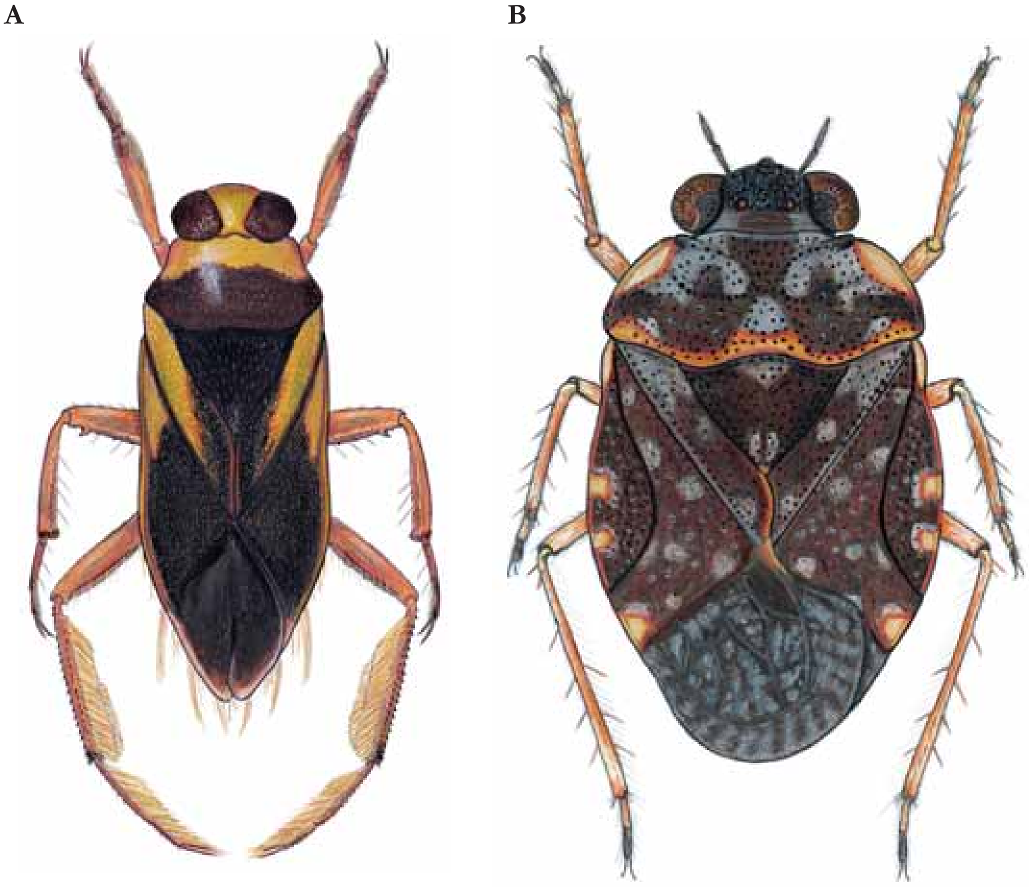

Figure A3.11 A. Notonectidae. *Notonecta obliqua.* (x8) **B. Ochteridae.** *Ochterus marginatus marginatus.* (x28)

Appendix IV

Key to the tribes of Miridae from the Iberian Peninsula

Mature stages

Macropterous, brachypterous and micropterous forms

1. Ocelli present. Tarsi 2-segmented. [Figure A4.1A] .... **Isometopini** Fieber, 1860
One species: *Isometopus intrusus* (Herrich-Schaeffer, 1835)

— Ocelli absent. Tarsi 3-segmented **2**

2. Pronotum distinctly divided in two transversal lobes, the posterior at least 2.5 times longer than the anterior [Figure A4.1B] .... **Nichomachini** Schuh, 1974
Two species: *Laurinia camponotidea* (Lindberg, 1940)
Laurinia fugax Reuter, 1884

— Pronotum not divided in transversal lobes[a] **3**

3. Antenomers II are the longest antennal segments and tarsomers III are the longest tarsal segments. [Figure A4.2] .... **Bryocorini** Baerensprung, 1860

— Antenomers II are not the longest antennal segments and/or tarsomers III are not the longest tarsal segments **4**

4. Anterior region of pronotum presenting a longitudinally-bilobed callosity and antenomers I and II thicker than antenomers III and IV. [Figure A4.3] .... **Fulviini** Uhler, 1886
Two species: *Fulviuis borgesi* Cherot, Ribes & Gorczyca, 2006
Fulvius oxycarenoides Reuter, 1878

— Anterior region of pronotum without a longitudinally-bilobed callosity and/or antenomers I and II aprox. equal or thinner than antenomers III and IV **5**

5. Ungitractor plate of claws presenting membranaceous parempodia **6**

— Ungitractor plate of claws presenting setiform parempodia **10**

6. Apexes of parempodia diverging from each other **7**

— Apexes of parempodia converging to each other **8**

7. Metatarsomers I aprox. equal or longer than metatarsomers II and III taken together. Macro- or brachypterous. [Figure A4.4] .... **Stenodimini** China, 1943

— Metatarsomers I shorter than metatarsomers II and III taken together. Always macropterous. [Figure A4.5] .... **Mirini** Hahn, 1833

8. As measured in lateral view, the distance from the apex of head to the margin of the eye is longer than the eye's longest axis. [Figure A4.6] . . . . **Halticini** Costa, 1853

— As measured in lateral view, the distance from the apex of head to the margin of the eye is aprox. equal or shorter than the eye's longest axis **9**

9. Posterior margin of head covering pronotal anterior margin. Antenomers I presenting spines. Always macropterous. [Figure A4.7] . . . . . **Pilophorini** Douglas and Scott, 1876

— Posterior margin of head not covering pronotal anterior margin. Antenomers I without spines. Macro- or brachypterous. [Figure A4.8] . . . . **Orthotylini** Van Duzee, 1916

10. Pronotum presenting collar **11**

— Ponotum without collar. [Figure A4.9] . . . . **Phylini** Douglas and Scott, 1865

11. Dorsum of body punctuated **12**

— Dorsum of body without punctuations **13**

12. Anterior region of pronotum presenting a transversal furrow. Macropterous forms present hairy membranes. Macro- or brachypterous. [Figure A4.10] . . . . **Clivinematini** Reuter, 1876

One species: *Bothynotus pilosus* (Boheman, 1852)

— Anterior region of pronotum without transversal furrow. Membranes without hairs. Always macropterous. [Figure A4.11] . . . . . . **Deraeocorini** Douglas and Scott, 1865

13. Scutellum pale-greenish unicolored, reddish and whitish bicolored, or dark with contrasting pale spots. [Figure A4.12] . . . . **Dicyphini** Reuter, 1883

— Scutellum dark-reddish, brownish or black uncolored. [Figure A4.13] . . . . **Hallodapini** Van Duzee, 1916

Notes

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a In a few species of the genus *Dicyphus* Fieber, 1858 (Dicyphini) the pronotum may appear to be divided in anterior and posterior lobes but these 'lobes' are of aprox. equal lengths.

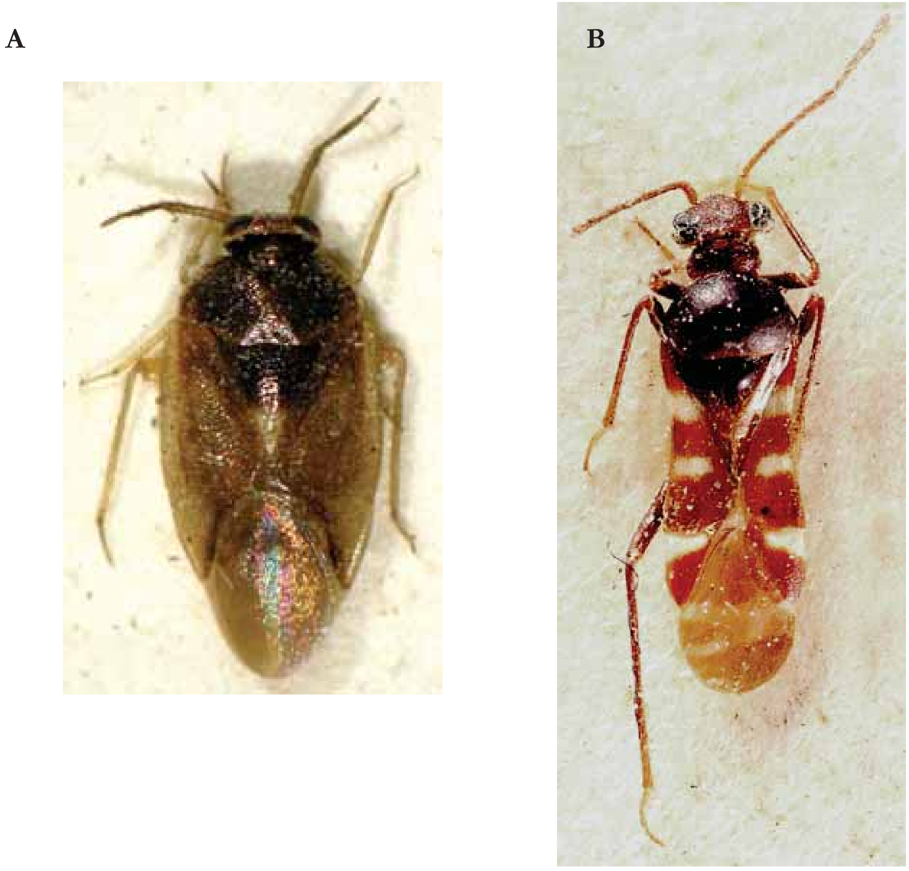

Figure A4.1 A. Isometopini *Isometopus intrusus* Source: Michael F. Schönitzer (Wikimedia Commons) **B. Nichomachini** *Laurinia fugax* Source: American Musuem of Natural History (Discoverlife)

Figure A4.2 Bryocorini *Monalocoris filicis* Source: Jospeh Lynn (Flickr)

Figure A4.3 Fulviini *Fulvius sp.* Source: Eric Gofreed (Flickr)

Figure A4.4 Stenodimini *Leptopterna dolabrata* Source: Jospeh Lynn (Flickr)

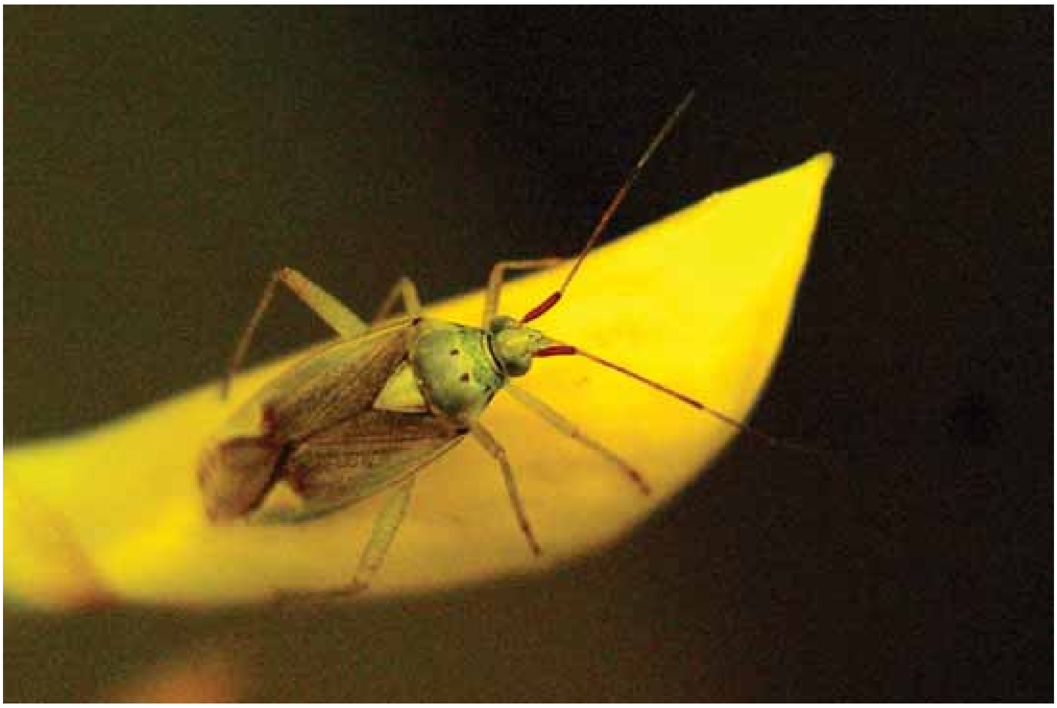

Figure A4.5 Mirini *Adelphocoris lineolatus* Source: Original

Figure A4.6 Halticini *Orthocephalus saltator* Source: Ian Boyd (Flickr: Creative Commons)

Figure A4.7 Pilophorini *Pilophorus perplexus* Source: Ian Boyd (Flickr: Creative Commons)

Figure A4.8 Orthotylini *Orthotylus marginalis* Source: Joseph Lynn (Flickr)

Figure A4.9 Phylini *Amblytylus nasutus* Source: Joseph Lynn (Flickr)

Figure A4.10 Clivinematini *Bothynotus pilosus* Source: Petri Parkko

Figure A4.11 Deraeocorini *Deraeocoris lutescens* Source: Tristan Bantock (Flickr)

Figure A4.12 Dicyphini *Dicyphus epilobii* Source: Joseph Lynn (Flickr)

Figure A4.13 Hallodapini *Cremnocephalus albolineatus* Source: Arto Muinonen (Flickr)

Appendix V.1 Key to the genera of Rhyparochrominae from the Iberian Peninsula

Mature stages

Macropterous forms

1. Suture between sterna IV-V approx. straight, reaching or almost reaching abdominal lateral margins **2**

— Suture between sterna IV-V curving towards the base of abdomen, not reaching the abdominal lateral margin **3**

2. Pronotal posterior margin narrower than widest region of abdomen. Pronotal lateral margin laminar. Diatone approx. as wide as pronotal anterior margin. Eyes separated from pronotum. Membranes hyaline. [Figure A5.1] . . . . *Gastrodes* Westwood, 1840

— Pronotal posterior margin approx. as wide as widest region on abdomen. Pronotal lateral margin rounded, presenting a sharp edge. Diatone wider than pronotal anterior margin. Eyes approx. in contact with pronotum. Membranes colored. [Figure A5.2] . . . . *Acompus* Fieber, 1860

3. Body length between 1.5 and 1.7 mm . . . . *Stygnocorisella* Hoberlandt, 1956
One species: *Stygnocorisella mayeti* (Puton, 1879)

— Body length longer than 1.9 mm **4**

4. Pronotum presenting an impunctured 3-pointed black pattern against a punctured pale background. [Figure A5.3] . . . . *Lasiosomus* Fieber, 1860
One species: *Lasiosomus enervis* (Herrich-Schaeffer, 1835)

— Pronotum uniformly monocolored or variously patterned, never with an inverse impunctured 3-pointed black crown against a punctured pale background **5**

5. Pronotum presenting a post-median transversal furrow that reaches all the way to the ventrum of prothorax **6**

— Pronotum presenting a post-median transversal furrow that reaches at most the pronotal lateral margins or pronotum without transversal furrow **10**

6. Pronotum presenting collar **7**

— Pronotum without collar **8**

7. Antenomer IV dark. Tarsomers III pale. Body length between 4.7 and 5.7 mm. [Figure A5.4] . . . . *Pachybrachius* Hahn, 1826
One species: *Pachybrachius fracticollis* (Schilling, 1829)

— Antenomer IV pale to reddish. Tarsomers III dark. Body length between 6.5 and 8.3 mm. [Figure A5.5] . . . . *Paromius* Fieber, 1860
One species: *Paromius gracilis* (Rambur, 1839)

8. Posterior region of pronotum testaceous. Body length between 5.0 and 6.0 mm .... *Remaudiereana* Hoberlandt, 1954
One species: *Remaudiereana annulipes* (Baerenssprung, 1859)
— Posterior region of pronotum almost entirely dark. Body length between 5.1 and 7.2 mm **9**
9. Antenomer II pale and dark bicolored. Antenomer IV uniformly dark. [Figure A5.6] . ... *Ligyrocoris* Stål, 1872
One species: *Ligyrocoris sylvestris* (Linnaeus, 1758)
— Antenomer II uniformly pale. Antenomer IV pale and dark bicolored .... *Paraparomius* Harrington, 1980
One species: *Paraparomius leptopoides* (Baerensprung, 1859)
10. Pronotal lateral margins <u>and</u> anterior region of hemelytral lateral margins presenting a fringe of long erect hairs. [Figure A5.7] .... *Pionosomus* Fieber, 1860
One species: *Pionosomus varius* (Wolff, 1804)
— Pronotal lateral margins <u>and/or</u> anterior region of hemelytral lateral margins without a fringe of long erect hairs **11**
11. Pronotal lateral margins rounded, presenting or not sharp edges **12**
— Pronotal lateral margins laminar **17**
12. Body length between 2.0 and 4.2 .... **13**
— Body length between 4.8 and 7.6 .... **16**
13. Head and scutellum yellowish to light-brownish. Body length between 2.0 and 2.5 mm .... *Camptocera* Jakovlev, 1877
One species: *Camptocera glaberrima* (Walker, 1872)
— Head and scutellum dark. Body length between 2.4 and 4.2 mm **14**
14. Buccula joining at a point towards the base of head .... *Tropistethus* Fieber, 1860
— Buccula do not join towards the base of head **15**
15. Pronotal posterior margin dark. Antennae uniformly dark. Profemora without spines. Body length between 2.5 and 3.2 mm .... *Aoploscelis* Fieber, 1860
One species: *Aoploscelis bivirgata* (Costa, 1835)
— Pronotal posterior margin whitish. Antennae pale and dark bicolored. Profemora presenting at least one large spine. Body length between 3.2 and 4.2 mm .... *Icus* Fieber, 1860
One species: Icus angularis Fieber, 1861
16. Coria pale or transversely pale and dark bicolored. Legs dark. Body length between 4.8 and 5.8 mm .... *Pterotmetus* Amyot and Serville, 1843
One species: *Pterotmetus staphyliniformis* (Schilling, 1829)
— Coria dark. Legs light-reddish. Body length between 6.2 and 7.6 mm .... *Proderus* Fieber, 1860
One species: *Proderus suberythropus* (Costa, 1842)
17. Scutellum pale to testaceous **18**
— Scutellum pale and dark bicolored or entirely dark **21**
18. Anterior angles of pronotum projected outwards and forward **19**
— Anterior angles of pronotum not projected outwards and forward **20**

Figure A5.1 *Gastrodes abietum* Source: Guido Bohne

Figure A5.2 *Acompus rufipes* Source: Roland Kratzer

19. Pronotal anterior margin distinctly concave. Body length between 4.5 and 8.2 mm. [Figure A5.8] .... *Emblethis* Fieber, 1860

— Pronotal anterior margin approx. straight. Body length between 4.4 and 5.5 mm .... *Gonianotus* Fieber, 1860

20. Antennae approx. as wide as apex of metatibia. Body length between 1.9 and 3.6 mm .... *Notochilus* Fieber, 1860

— Antennae distinctly narrower than apex of metatibia. Body length between 8.5 and 10.0 mm .... *Lethaeus* Dallas, 1852

One species: *Lethaeus fulvovarius* Puton, 1884

21. Head, pronotum, scutellum, clavus and coria entirely dark. Body length between 6.5 and 10.0 mm **22**

— Head, pronotum, scutellum, clavus or coria not entirely dark. Body length between 2.3 and 11.0 mm **25**

22. Dorsum of body presenting hairs .... *Microtomideus* Reuter, 1885

— Dorsum of body without hairs **23**

23. Antennae, tibia and tarsi light-reddish. Body length between 7.5 and 10.0 mm .... *Neurocladus* Fieber, 1860

One species: *Neurocladus brachiidens* (Dufour, 1851)

— Antennae, tibia and tarsi dark. Body length between 6.5 and 9.0 mm **24**

24. Membranes dark, frequently presenting a large whitish to reddish spot towards their base. Body length between 6.5 and 7.5 mm. [Figure A5.9] .... *Aphanus* Laporte, 1833

One species: *Aphanus rolandri* (Linnaeus, 1758)

— Membranes dark, frequently presenting paler regions between the veins. Body length between 8.3 and 9.0 mm .... *Aellopus* Wolff, 1811

One species: *Aellopus atratus* (Goeze, 1778)

25. Scutellum dark, presenting towards its apex some form of contrasting pale mark **26**

— Scutellum entirely dark **33**

26. Diatone wider than pronotal anterior margin. Body length between 2.3 and 7.6 mm **27**

— Diatone approx. equally wide or narrower than pronotal anterior margin. Body length between 6.4 and 11.0 mm **30**

27 Endocoria presenting towards its posterior region a large roundish pale spot that extends to the external margin of exocoria. [Figure A5.10] .... *Beosus* Amyot and Serville, 1843

— Endocoria without large roundish pale spot **28**

28. Profemora presenting spines. Body length between 3.4 and 6.0 mm **29**

— Profemora without spines. Body length between 2.3 and 3.1 mm .... *Ischnocoris* Fieber, 1860

Figure A5.3 *Lasiosomus enervis* Source: Tristan Bantock

Figure A5.4 *Pachybrachius fracticollis* Source: Tristan Bantock

29. Anterior region of pronotum presenting a dark transversal band that covers at most half of the pronotal surface. Transversal dark band of pronotum not reaching the pronotal lateral margins. Membranes hyaline. Body length between 3.4 and 4.0 mm . . . . *Hyalochilus* Fieber, 1860
One species: *Hyalochilus ovatulus* (Costa, 1853)

— Anterior region of pronotum presenting a dark transversal band that covers at least half of the pronotal surface. Transversal dark band of pronotum reaching the pronotal lateral margins. Membranes colored. Body length between 4.0 and 6.0 mm. [Figure A5.11] . . . . *Peritrechus* Fieber, 1860

30. Femora pale and dark bicolored. [Figure A5.12] . . . . *Dieuches* Dohrn, 1860
One species: *Dieuches armatipes* (Walker, 1872)

— Femora entirely dark **31**

31. Pronotal lateral margins slightly convex. Dark spot of the posterior region of coria roundish. Posterior margin of membranes entirely pale . . . . *Lasiocoris* Fieber, 1860

— Pronotal lateral margins slightly concave. Dark spot of the posterior region of coria rhomboidal or irregularly-shaped. Posterior margin of membranes never entirely pale **32**

32. Posterior region of coria presenting a pale spot. Body length between 6.8 and 8.0 mm . [Figure A5.13] . . . . *Graptopeltus* Stål, 1872
One species: *Graptopeltus lynceus* (Fabricius, 1775)

— Posterior region of coria without a pale spot. Body length between 9.0 and 10.0 mm . . . . *Trichaphanus* Kiritschenko, 1926
One species: *Trichaphanus fuentei* (Puton, 1894)

33. Posterior region of coria dark, presenting a large roundish pale spot. Body length between 3.0 and 3.2 mm. . . . . Tempyra Stål, 1874
One species: *Tempyra biguttula* Stål, 1874

— Posterior region of coria pale to dark, never presenting a large roundish pale spot. Body length between 2.6 and 8.7 mm **34**

34. Pronotal anterior and posterior margins approx. equally wide. Body length between 3.2 and 4.0 mm . . . . *Macrodema* Fieber, 1860
One species: *Macrodema micropterum* (Curtis, 1836)

— Pronotal anterior margin distinctly narrower than posterior margin. Body length between 2.6 and 8.7 mm **35**

35. Profemora presenting at least one spine. Body length between 2.9 and 8.7 mm **36**

— Profemora without spines. Body length between 2.6 and 4.5 mm. [Figure A5.14] . . . . *Stygnocoris* Douglas and Scott, 1865

36. Coria entirely pale. Body length between 3.1 and 4.0 mm . . . . *Thaumastopus* Fieber, 1870
One species: *Thaumastopus marginicollis* (Lucas,1849)

— Coria never entirely pale. Body length between 2.9 and 8.7 mm **37**

37. Membranes whitish, presenting towards their centroid a large black spot **38**

— Membranes hyaline, whitish, dark or bicolored, never presenting towards their centroid a large black spot **39**

Figure A5.5 *Paromius gracilis* Source: Endika Ussia

Figure A5.6 *Ligyrocoris sylvestris* Source: Tom Murray

38\. Endocoria dark. Pronotum almost entirely dark. Body length between 3.5 and 4.6 mm . . . . *Lamprodema* Fieber, 1860
One species: *Lamprodema maura* (Fabricius, 1803)

— Endocoria pale and dark bicolored. Pronotum pale and dark bicolored. Body length between 4.5 and 8.4 mm. [Figure A5.15] . . . . *Xanthochilus* Stål, 1872

39\. Antenomer I reaching at most the apex of clypeus. Body length between 3.1 and 5.0 mm **40**

— Antenomer I overreaching the apex of clypeus. Body length between 2.9 and 8.7 mm **41**

40\. Pronotum dark. Coria dark. Protibia arched. Body length between 3.1 and 4.0 mm . . . . *Piezoscelis* Fieber, 1870
One species: *Piezoscelis staphylinus* (Rambur, 1839)

— Pronotum pale and dark bicolored. Coria yellowish to reddish, presenting towards its posterior region a roundish brownish spot. Protibia straight. Body length between 4.7 and 5.5 mm . . . . *Pezocoris* Jakovlev, 1875
One species: *Pezocoris apicimacula* (Costa, 1853)

41\. Eyes separated from the pronotal anterior margin. Body length between 2.9 and 8.0 mm **42**

— Eyes approx. in contact with the pronotal anterior margin. Body length between 3.5 and 8.7 mm **45**

42\. Head and anterior region of pronotum densely punctuated. Body length between 3.0 and 6.5 mm **43**

— Head and anterior region of pronotum barely punctuated. Body length between 2.9 and 8.0 mm **44**

43\. Antenomer I overreaching the apex of clypeus by more than half of its length. Pronotum presenting a narrow collar. Body length between 3.2 and 4.6 mm. [Figure A5.16] . . . . *Taphropeltus* Stål, 1872

— Antenomer I overreaching the apex of clypeus by at most half of its length. Pronotum without a collar. Body length between 3.0 and 6.5 mm. [Figure A5.17] . . . . *Drymus* Fieber, 1860

44\. Antenomer I overreaching the apex of clypeus by more than half of its length. Pronotum without a collar. Body length between 4.9 and 8.0 mm. [Figure A5.18] . . . . *Eremocoris* Fieber, 1860

— Antenomer I overreaching the apex of clypeus by at most half of its length. Pronotum presenting a narrow collar. Body length between 2.9 and 4.7 mm. [Figure A5.19] . . . . *Scolopostethus* Fieber, 1860

45\. Posterior region of pronotum dark. [Figure A5.20] . . . . *Megalonotus* Fieber, 1860

— Posterior region of pronotum not entirely dark **46**

46\. Laminar expansion of the pronotal lateral margins approx. equally wide from its anterior to posterior region. Body length between 3.8 and 6.2 mm. [Figure A5.21] . . . . *Trapezonotus* Fieber, 1860

— Laminar expansion of the pronotal lateral margins distinctly wider towards its middle region. Body length between 5.2 and 8.7 mm **47**

Figure A5.7 *Pionosomus varius* Source: Tristan Bantock

Figure A5.8 *Emblethis griseus* Source: Tristan Bantock

47. Metafemora presenting at least one spine. Body length between 5.2 and 7.7 mm . . . . *Raglius* Stål, 1872

— Metafemora without spines. Body length between 6.8 and 8.7 mm. [Figure A5.22] . . . . *Rhyparochromus* Hahn, 1826

Figure A5.9 *Aphanus rolandri* Source: Endika Ussia

Figure A5.10 *Beosus maritimus* Source: Endika Ussia

Figure A5.11 *Peritrechus geniculatus* Source: Rob Ryan

Figure A5.12 *Dieuches armatipes* Source: Gail Hampshire

Figure A5.13 *Graptopeltus lynceus* Source: Rob Ryan

Figure A5.14 *Stygnocoris sabulosus* Source: Tom Murray

Figure A5.15 *Xanthochilus saturnius* Source: Original.

Figure A5.16 *Taphropeltus contractus* Source: Rob Ryan

Figure A5.17 *Drymus reyi* Source: Rob Ryan

Figure A5.18 *Eremocoris fenestratus* Source: Tristan Bantock

Figure A5.19 *Scolopostethus thomsoni* Source: Tom Murray

Figure A5.20 *Megalonotus sabulicola* Source: Tom Murray

Figure A5.21 *Trapezonotus desertus* Source: Rob Ryan

Figure A5.22 *Rhyparochromus pini* Source: Rob Ryan

Appendix V.2

Key to the genera of Rhyparochrominae from the Iberian Peninsula

Mature stages

Brachypterous forms

1. Suture between sterna IV-V approx. straight, reaching or almost reaching abdominal lateral margins. Body length between 3.5 and 4.5 mm .... *Acompus* Fieber, 1860
— Suture between sterna IV-V curving towards the base of abdomen, not reaching the abdominal lateral margin. Body length between 1.9 and 8.0 mm **2**
2. Body length between 6.2 and 8.0 mm **3**
— Body length between 1.9 and 6.0 mm **4**
3. Pronotum and coria pale and dark bicolored. Dorsum of connexiva dark .... *Ragliodes* Reuter, 1885
One species: *Ragliodes delineates* (Rambur, 1839)
— Pronotum and coria testaceous. Dorsum of connexiva pale and dark bicolored .... *Ischnopeza* Fieber, 1860
One species: *Ischnopeza hirticornis* (Herrich-Schaeffer, 1850)
4. Dorsum of abdomen and connexiva pale to testaceous. Body length between 2.0 and 2.5 mm .... *Camptocera* Jakovlev, 1877
One species: *Camptocera glaberrima* (Walker, 1872)
— Dorsum of abdomen and connexiva dark. Body length between 1.9 and 6.0 mm **5**
5. Pronotal lateral margins rounded. Buccula jointed at a point between the eyes and the base of the head .... *Tropistethus* Fieber, 1860
One species: *Tropistethus pallipes* Reuter, 1902
— Pronotal lateral margins roundish with sharp edges <u>or</u> margins laminar. Buccula not jointed **6**
6. Pronotal lateral margins roundish with sharp edges <u>and</u> posterior margin of tergite V concavely curved **7**
— Pronotal lateral margins laminar <u>and/or</u> posterior margin of tergite V straight ... **8**
7. Coria almost entirely dark. Profemora without spines. Body length between 2.5 and 3.2 mm .... *Aoploscelis* Fieber, 1860
One species: *Aoploscelis bivirgata* (Costa, 1835)
— Coria pale <u>or</u> transversely pale and dark bicolored. Profemora presenting one or more spines. Body length between 4.8 and 5.8 mm ...... *Pterotmetus* Amyot and Serville, 1843
8. Diatone wider than pronotal anterior margin. Pronotum without collar. Body length between 2.3 and 4.6 mm **9**
— Diatone approx. equally wide as pronotal anterior margin. Pronotum presenting or not collar. Body length between 1.9 and 6.0 mm **11**

9. Pronotal anterior and posterior margins approx. equally wide. Scutellum pale and dark bicolored. Coria pale . . . . *Ischnocoris* Fieber, 1860

— Pronotal anterior margin narrower than posterior margin. Scutellum dark. Coria testaceous to dark **10**

10. Profemora presenting one spine. Pronotal lateral margins slightly concave. Clavus generally lighter in color than coria . . . . *Lamprodema* Fieber, 1860

One species: *Lamprodema maura* (Fabricius, 1803)

— Profemora without spines. Pronotal lateral margins slightly convex. Clavus and coria of the same color . . . . *Stygnocoris* Douglas and Scott, 1865

11. Coria dark, presenting contrasting pale lines running parallel to the lateral margins of scutellum. Body length between 3.2 and 3.8 mm . . . . *Hispanocoris* Costas and Vázquez, 1999

One species: Hispanocoris pericarti Costas & Vázquez, 1999

— Coria pale to dark, monocolored or bicolored, never presenting contrasting lines running parallel to the lateral margins of scutellum. Body length between 1.9 and 6.0 mm **12**

12. Pronotum dark. Body length between 3.1 and 5.5 mm **13**

— Pronotum bicolored. Body length between 1.9 and 6.0 mm **14**

13. Head as long as wide. Anterior region of pronotum approx. 3 times as long as posterior region. Posterior margin of tergite V concavely curved. Clavus lighter in color than endocoria. Body length between 3.1 and 4.0 mm . . . . *Piezoscelis* Fieber, 1870

One species: *Piezoscelis staphylinus* (Rambur, 1839)

— Head wider than larger. Anterior region of pronotum at most 2 times as long as posterior region. Posterior margin of tergite V straight. Clavus and endocoria of the same color. Body length between 4.0 and 5.5 mm . . . . Megalonotus *Megalonotus* Fieber, 1860

One species: *Megalonotus antennatus* (Schilling, 1829)

14. Latero-posterior angles of pronotum dark **15**

— Latero-posterior angles of pronotum pale to testaceous **16**

15. Anterior region of the pronotal lateral margins pale. Pronotum presenting collar. Eyes presenting hairs. Body length between 2.9 and 4.7 mm . . . . . *Scolopostethus* Fieber, 1860

— Anterior region of the pronotal lateral margins dark. Pronotum without collar. Eyes without hairs. Body length between 4.5 and 6.0 mm . . . . Raglius *Raglius* Stål, 1872

One species: *Raglius alboacuminatus* (Goeze, 1778)

16. Profemora at least as twice as thick as metafemora. Body length between 1.9 and 3.6 mm . . . . *Notochilus* Fieber, 1860

— Profemora at most slightly thicker than metafemora. Body length between 3.2 and 5.1 mm **17**

17. Pronotal anterior and posterior margins approx. equally wide. Pronotal anterior margin straight. Profemora without spines. Body length between 3.2 and 4.0 mm . . . . *Macrodema* Fieber, 1860

One species: *Macrodema micropterum* (Curtis, 1836)

— Pronotal anterior margin narrower than posterior margin. Pronotal anterior margin slightly concave. Profemora presenting spines. Body length between 3.6 and 5.1 mm . . . . *Trapezonotus* Fieber, 1860

Appendix VI

Key to the species of *Deraeocoris* from the Iberian Peninsula

Mature stages

1. Antenomer I distinctly longer than diameter of eye. Body length between 5.1 and 12.0 mm **2**

— Antenomer I approx. equally long as diameter of eye. Body length between 3.4 and 5.3 mm **9**

2. Apex of antenomer II distinctly thicker than its base **3**

— Apex of antenomer II approx. equally wide as its base **4**

3. Collar of pronotum glossy. Body length between 6.5 and 7.7 mm. [Figure A6.1] . . . . *Deraeocoris ruber* (Linnaeus, 1758)

— Collar of pronotum matte. Body length between 5.5 and 6.2 mm. [Figure A6.2] . . . . *Deraeocoris punctum* (Rambur, 1839)

4. Pronotum and hemelytra presenting long erect hairs . . . . *Deraeocoris schach* (Fabricius, 1781)

— Pronotum and hemelytra without hairs **5**

5. Body length between 8.5 and 12.0 mm . . . . *Deraeocoris trifasciatus* (Linnaeus, 1767)

— Body length between 5.4 and 7.8 mm **6**

6. Collar of pronotum pale, yellowish-brownish or reddish. Body length between 5.5 and 7.0 mm. [Figure A6.3] . . . . *Deraeocoris flavilinea* (Costa, 1862)

— Collar of pronotum black. Body length between 5.4 and 7.8 mm **7**

7. Exocoria entirely yellowish to orange or yellowish (anteriorly) and black (posteriorly) bicolored. Body length between 5.4 and 6.4 mm. [Figure A6.4] . . . . *Deraeocoris cordiger* (Hahn, 1834)

— Exocoria entirely black. Body length between 5.5 and 7.8 mm **8**

8. Openings of the metathoracic scent-glands entirely pale. Body length between 5.5 and 6.6 mm. [Figure A6.5] . . . . *Deraeocoris ribauti* Wagner, 1943

— Openings of the metathoracic scent-glands at least partially dark. Body length between 6.0 and 7.8 mm . . . . *Deraeocoris morio* (Boheman, 1852)

9. Scutellum presenting punctures. Body length between 3.4 and 4.0 mm . . . . *Deraeocoris serenus* Douglas & Scott, 1868

— Scutellum without punctures. Body length between 3.8 and 5.3 mm **10**

10. Diatone wider than anterior margin of pronotum. Eyes distinctly separated from pronotum. Body length between 4.0 and 5.3 mm . . . . . *Deraeocoris martini* (Puton, 1887)

— Diatone approx. equally wide as anterior margin of pronotum. Eyes approx. in contact with pronotum. Body length between 3.8 and 4.6 mm. [Figure A6.6] . . . . *Deraeocoris lutescens* (Schilling, 1837)

Figure A6.1 *Deraeocoris ruber.* Source: original.

Figure A6.2 *Deraeocoris punctum.* Source: Jessica Joachim.

Figure A6.3 *Deraeocoris flavilinea.* Source: Ian Boyd (Flickr).

Figure A6.4 *Deraeocoris cordiger.* Sorce: Costán Escuer (Flickr).

Figure A6.5 *Deraeocoris ribauti*. Source: Carlos Castañeda (Flickr).

Figure A6.6 *Deraeocoris lutescens*. Source: Tristan Bantock (Flickr).

Suplementary materials

Supplementary data and codes associated with this thesis can be found in the CD version:

Tables M. Additional tables associated to the 'Materials and methods' section (Tables M1 to M3)

Tables R. Additional tables associated to the 'Results' section (Tables R1 to R17)

Models. R and OpenBUGS codes for the multi-species site occupancy models (Models 1 to 7)

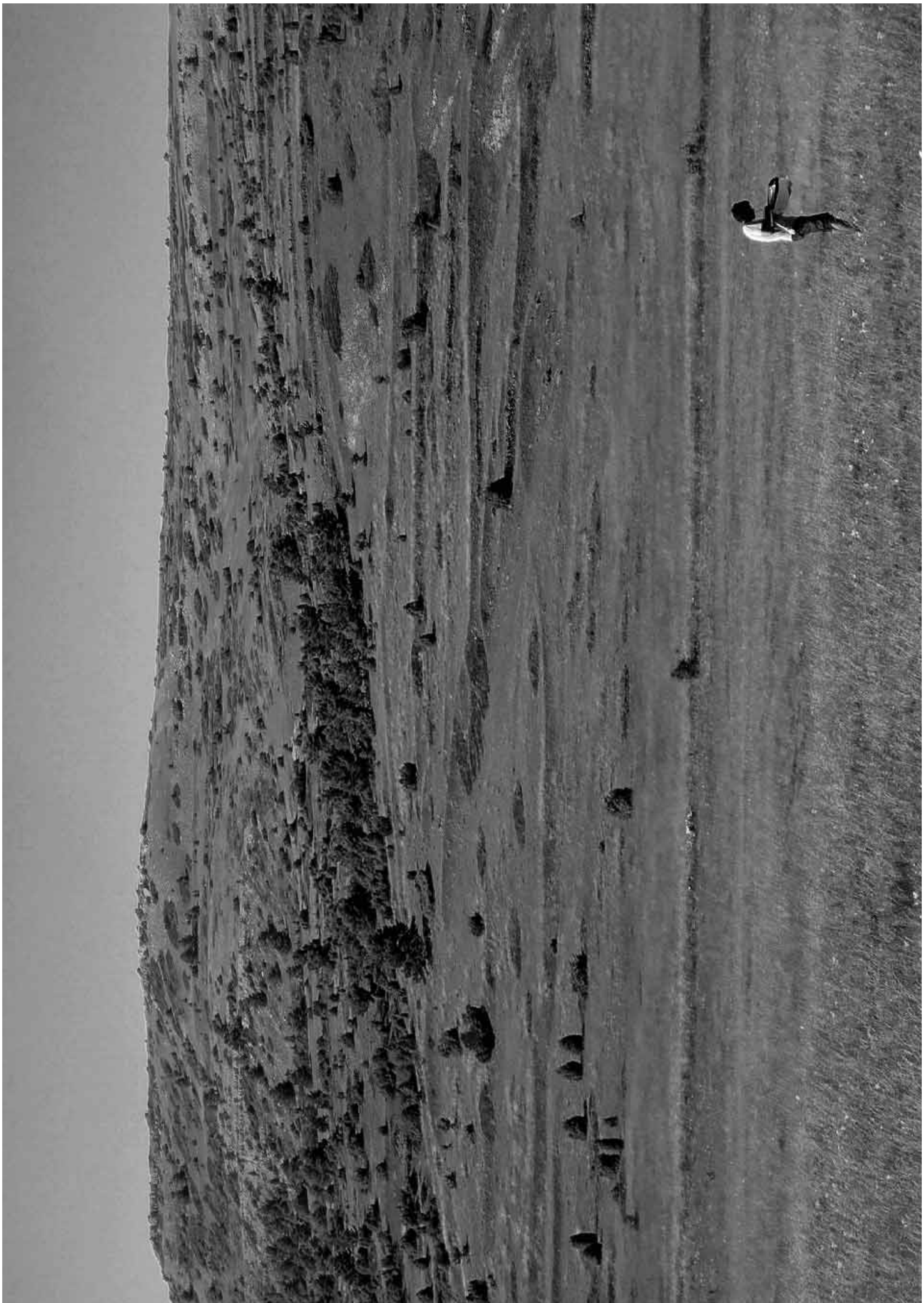

photo by der Vater