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Butterfly Diversity in a Changing Scenario

Simona Bonelli et al.*
University of Turin – Department of Animal and Human Biology, Turin, Italy

1. Introduction

For many historical and ecological reasons, Italy is characterized by extremely high biodiversity level, which we can observe in virtually all animal and plant groups. This occurs in concomitance with relatively low levels of anthropic disturbance, at least in comparison to most areas of central Europe. The elongate shape and mountainous nature of Italy, together with its vast littoral areas, contribute to generate very diverse ecological conditions, all along a series of latitudinal and altitudinal habitat gradients. The Mediterranean basin, where Italy has centre-stage position, is among the 25 world biodiversity hotspots identified in Myers' seminal paper (Myers et al., 2000). Whatever will happen to, or affect the, Italian biodiversity will be reflected on a world-wide scale. This chapter describes animal diversity as a dynamic functional system, now increasingly threatened by various kinds of anthropogenic threats and pressures. We describe how and to what extent biodiversity is changing, taking as an example a particularly well-known animal group, such as butterflies, within an area of focal importance, such as the Mediterranean basin. We mainly discuss data that we, and our research group, have personally obtained in the course of the past several years, and even though we mainly worked in Italy, we discuss them in the more general framework of a pan-European scientific scenario.

2. The Italian butterfly biodiversity

The Italian biodiversity is among the richest in Europe. In particular, the Italian butterfly fauna includes 283 species, assigned to 79 genera and 9 families or subfamilies. It represents 37% of the total euro-mediterranean fauna. At a national level, butterfly biodiversity is higher in northern Italy, particularly in the Alps and the pre-Alps, than in the Apennines and in the main Italian islands: this is a consequence of the well-known peninsular effect (Tontini et al., 2003), which develops along a latitudinal and longitudinal gradient, were the alpine arc plays a central role. Species richness notably differs also according to an altitudinal gradient. Forty-seven species are alpine elements present exclusively or primarily over the tree line boundaries, and ninety-seven are typical of the high mountain horizon. A smaller number of species (*Colias hyale, Maculinea alcon, M. teleius, Euphydryas aurinia,*

* Francesca Barbero, Luca P. Casacci, Cristiana Cerrato, Dario Patricelli, Marco Sala, Alessio Vovlas, Magdalena Witek and Emilio Balletto
University of Turin – Department of Animal and Human Biology, Turin,Italy

Coenonympha oedippus) live exclusively in the Padano-Venetian plains, whilst twelve species are elements connected with the Mediterranean woodlands (macchia).

The Italian endemic species *sensu stricto* are eighteen, i.e. 7.5% of the total, but this number would increase to 48 (17,1% of total), if we consider endemic species ranging across a small area of politically non-Italian territory, such as *Erebia christi* or *E. flavofasciata*.

The highest concentration of strictly Italian endemics is found in the Apennine mountains (10 species), in the small islands (4), Sardinia (3), Sicily (2) and in the xerothermic "oases" of the western Alps. During the last decade, two species newly became part of the Italian fauna: *Danaus chrysippus* and *Cacyreus marshalli*. *Danaus chrysippus* is a well-known migrant and became naturalised in several Italian regions (Apulia, Sardinia, Sicily etc.), after having already marked some brief presence in previous times. *C. marshalli* was accidentally introduced and was observed in Italy (Rome) for the first time in 1997, but its expansion throughout the whole Country was very fast.

Another species, *Lycena helle*, probably became extinct in historical times. Three more species, *Araschnia levana*, *Euphydryas maturna* and *Polyommatus exuberans* once supposed to be extinct, have been rediscovered in recent times (Balletto et al., 2007).

The Italian Ministry for the Environment has published in 2007 a preliminary distribution atlas for 10,000 animal species, the CKmap Project, mapped on a 10x10 Km UTM grid (e.g. Fig. 1). Data included in this database derive from the literature as well as from museum collections. As concerns butterflies, this data set currently includes well over 160 000 non-duplicated individual records. New data represents over 50% of the total data and are continuously updated (Balletto et al., 2007).

2.1 Population extinctions

In a recent paper (Bonelli et al., 2011), we focused on a database of well-documented population extinctions having occurred among Italian day-flying Lepidoptera.

Depth studies of patterns of extinction are fundamental to understand species vulnerability, in particular when population extinctions are not driven by habitat loss, but related to subtle changes in habitat quality and are due to 'unknown causes'. We used a dataset containing over 160,000 non duplicate individual records of occurrence referred to 280 butterflies and 43 zygaenid moths, and their relative extinction data, to carry out a twofold analysis. The earliest published data that we could use to evaluate extinctions are due to Hübner (1790) and de Prunner (1798). We identified ecological preferences that influence extinction probability, and we analysed if all species were equally vulnerable to the same factors. We investigated Italian population extinctions at two levels, i.e. i) Population level: data of extinctions were pooled across species, separately for their ecological features, because we hypothesized that populations sharing similar ecological traits would react to changes in a similar way, acting like '*functional types*' (Henle, 2004; Shreeve et al., 2001; Thompson et al., 1996). We tried to understand, in this way, if different ecological requirements influenced the species' probability of becoming extinct; ii) Species level: Italian species were subdivided into two groups comprising a) species that lost at least one population, and b) species that did not lose any populations during our selected time frame. We analysed differences between groups, taking into account the ecological characteristics of individual species and trying to understand which features made them more prone to extinction. Although only one species (i.e. *Lycaena helle*) may have become extinct in Italy after or around 1925, a relatively high number of populations of day-flying Lepidoptera have disappeared. At
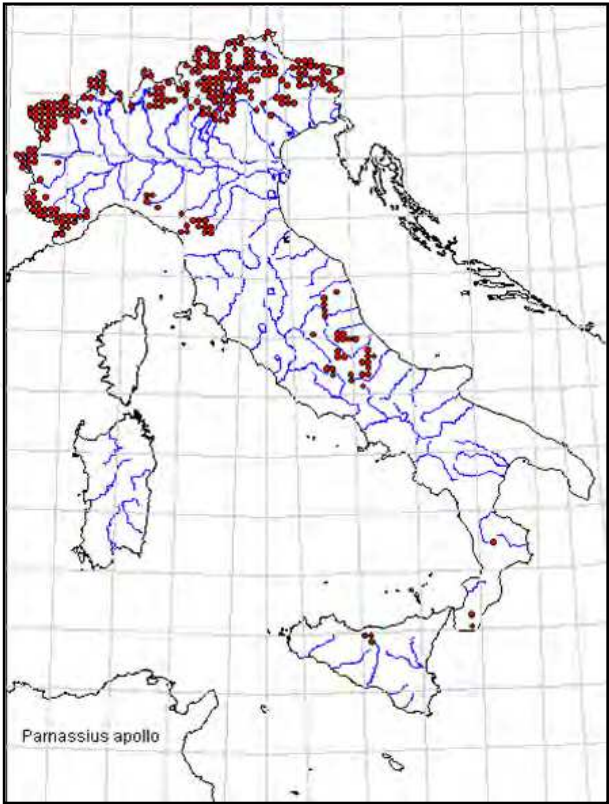
(a)

(b)

Fig. 1. Italian distribution of *Parnassius apollo* (a) (from Balletto et al., 2007); (b) *Parnassius apollo* courtesy of C. Bertino.

least 727 populations (653 of butterflies and 74 of burnet moths), formerly occurring in 268 UTM grid squares of the 3537 included in the Italian political territory have been lost. Population extinction affected 164 species, i.e. 142 butterflies (50.5% of the Italian fauna) and 22 zygaenids (51.2% of the Italian fauna). Our analyses revealed that extinctions were non-randomly distributed in space and time, as well as across species. Most of the extinctions were recorded in 1901-1950 and, as expected, populations at their range edges were more prone to become extinct for non-habitat related causes. Ecological traits were not only unequally distributed between extinction and non-extinction events, but also not all ecological features had the same importance in driving population vulnerability. Hygrophilous and nemoral species were the most likely to experience population losses and the most prone to disappear even when their habitats remained apparently unchanged.
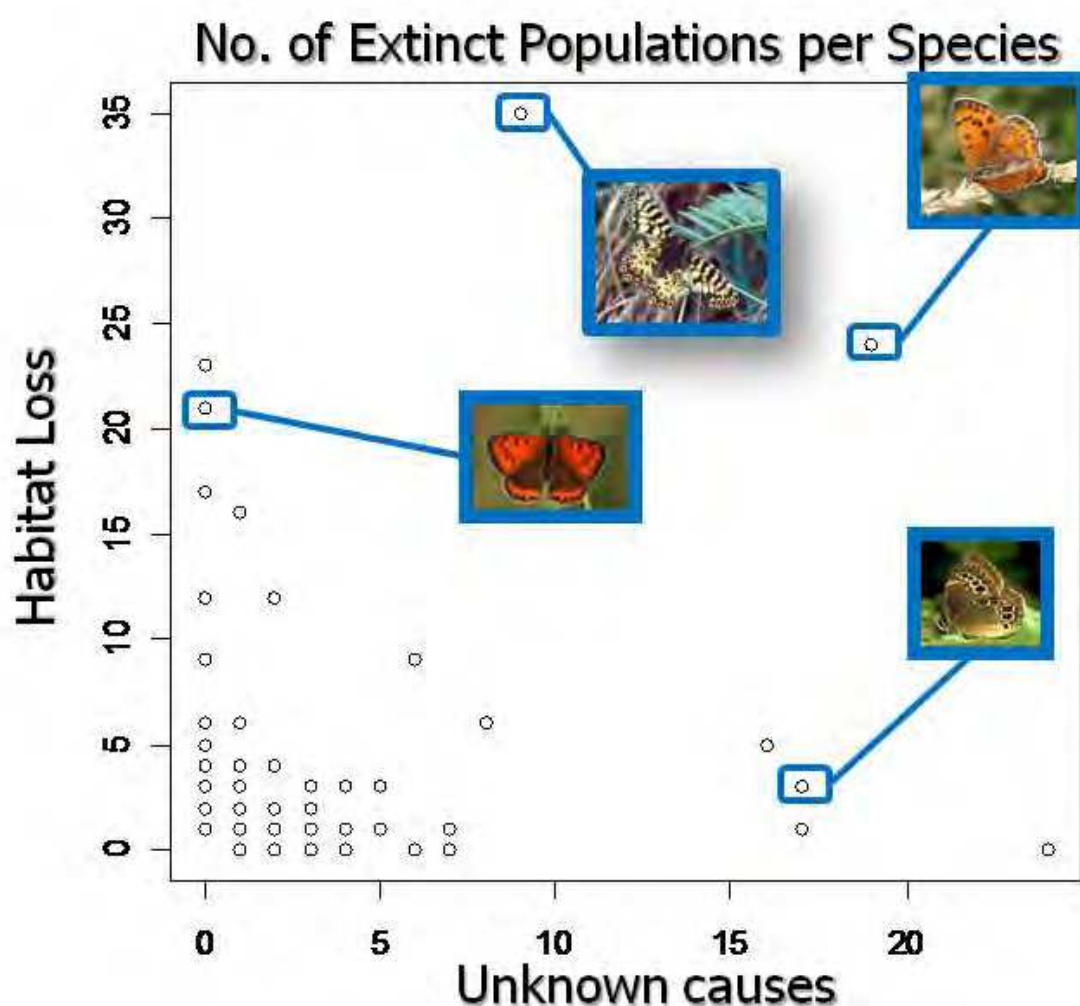

Fig. 2. Species showed a distinct pattern of vulnerability, depending on threats. Some species are more affected by unknown causes than by habitat loss (Unk. *vs.* Hab.) e.g. *Araschnia levana* (17 vs. 1), *Argynnis pandora* (16 vs. 5), *Melitaea britomartis* (24 vs. 0), *Lasiommata achine* (17 vs. 3, Fig. 2), whereas others are more affected by habitat loss, e.g. *Zerynthia polyxena* (9 vs. 35, Fig. 2), *Lycaena dispar* (0 vs. 21, Fig. 2), *Apatura ilia* (0 vs. 23), *Maculinea arion* (0 vs. 17), *Melanargia arge* (0 vs. 21). The only species that lost a high number of populations for both causes is *Lycaena thersamon* (19 vs. 24, Fig. 2).

As concern vulnerability to extinction, we tried to understand if different ecological traits, generating different ecological needs in ground water, light, temperature and general habitat preferences, influence extinction probability and vulnerability to different causes. It is, in fact, reasonable to postulate a non-random distribution of species' vulnerability (Isaac & Cowlishaw, 2004; McKinney, 1997; Parmesan, 2006). Species vulnerability depends on both ecological requirements and threat type: in fact, each species showed a distinct pattern of vulnerability, depending on threats (Fig. 2).

Habitat destruction is the main cause of extinction throughout the Italian territory, especially in the plains of the North, while hygrophilous and nemoral species are the most vulnerable. As already pointed out for many countries (e.g., van Swaay et. al., 2010) a correct conservation policy should begin by stopping urbanization and intensive agriculture and revitalising traditional agro pastoral activities. Our data, however, show that these measures may be insufficient. One-third of the Italian butterfly population extinctions, in fact, were not clearly related to habitat destruction, but linked to some more subtle degradation of environmental quality. Many, often still unknown, causes affect small, isolated populations. Van Swaay et al. (2006) indicated that butterflies are strongly declining all across Europe (-11% in the last 25 years) and that other threats, in particular global warming, will soon represent a serious matter of concern, as also indicated by Settele et al. (2008).

2.2 The extinction of Alpine species: The role of climate change

Climate studies strongly suggest that atmospheric alteration is already occurring and that changes in atmospheric composition are altering weather and climate processes. Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air temperatures (Ipcc, 2007). Temperature, however, is not the only climate variable likely to change. Widespread changes in precipitation amounts, wind patterns and aspects of extreme weather have already been observed (Ipcc, 2007). Closer investigation reveals that climatic change in the alpine region during the 20th century has been characterized by increases in minimum temperatures of up to 2°C, a more modest increase in maximum temperatures, little or no trend in the precipitation data, and a general decrease of sunshine duration, all through to about the mid-1980s (Beniston, 2000).

Given the central role of climate in governing the natural environment of mountain areas and the intensity of most biological processes, it is easy to suppose a high vulnerability of such systems to the impacts of a rapidly changing climate (Beniston, 2005). Mountain ecosystems provide interesting and useful models for the early detection and study of the signals of climatic change and its impacts on ecological systems (e.g., Haslett, 1997; Wilson et al., 2005).

Yet, any assessment of climatic change and of its related impacts in the mountains has been shown to be particularly difficult, because of the complexity of interrelated factors. In these regions topography is a dominant feature of the environment (Beniston, 2005) and determines rapid successions in various environmental conditions, even along narrow altitudinal gradients. Such difficulties are stronger in the Alps, which are characterised by diverse meso-climates even showing some Mediterranean influence, and for which it is almost impossible to draw up a comprehensive scenario of past and future changes valid for the entire alpine chain (Theurillat & Guisan, 2001). Moreover, the shape of landscapes has been modelled, during millennia, by human pressure and it is now difficult to understand the relative roles of land use changes and of global warming, or to hypothesize how these

factors will interact in the near future (e.g. Motta & Nola, 2001; Körner & Ohsawa, 2005; Vittoz et al., 2008).

To explore the relative role of climate and land use changes in shaping butterfly communities in mountain ecosystems, in 2009 we investigated a valley in the Maritime Alps (Italy, Valdieri, Valasco Valley), where we sampled 7 butterfly communities, already investigated in 1978. Sampling was made by semi-quantitative linear transects. Our 7 study sites had been precisely identified (spatially) in 1978, permitting the exact repetition of the monitoring programme in 2009. They range along an altitudinal gradient (1300–1900 m) and cover different kind of habitat (broadleaved forest, subalpine heathlands, hygrophilous meadows, alpine pastures, rocky slopes and screes). In 1978, the sampling period was limited to the end of July and the beginning of August, the optimal period to study butterfly communities in mountain ecosystems. In 2009 the sampling period was extended from the beginning of June until the end of August, to cover almost all the potential flight season, in order to be sure that '*no-more-found species*' really had disappeared during our time frame and had not just suffered some phenological shift.

Data from nearby meteorological stations, both rough and elaborated trough interpolation techniques (Loglisci, unpublished data), have been used to quantify climatic changes in the study area as a whole. Visual inspection of aerial photographs of each transect, taken in 1978 (source: IPLA archives, Piedmont) and repeated in 2006 (source: Italian Ministry for the Environment), have been used to describe changes in the vegetation cover during the selected time frames and to identify the most important habitat alterations.

The comparison of butterfly communities observed in 1987 and in 2009 did not evidence any clear pattern, in terms of species richness, between the two sampling periods, but the analysis of some individual cases provided interesting results, and showed several changes in the ecological composition patterns of butterfly communities.

We observed some '*species substitution*' in our sampling sites, indicating a general loss of specialised and narrow-range species and a general gain in ecologically tolerant elements. In 2009, we recorded neither *Coenonympha darwiniana*, an alpine endemic, nor *Coenonympha glycerion*, a hygrophilous species, but we found *Coenonympha arcania* in 6 of 7 sites, where it had not been recorded in 1978. The latter is a widespread species, characteristic of the lowland ecotonal habitats and much stronger generalist than the previous two. A similar case is that of *Colias phicomone*, a xerophilous species, linked to open herbaceous habitats occurring above the tree line (subalpine and alpine belts). It was repeatedly found in 1978, but was '*substituted*' in 2009 by the massive ingression of *Colias crocea*, an ecotonal, altitudinally generalist species, generally thermophilous and characterised by very high vagility.

Still at species level, thermal preferences demonstrated importance in determining patterns and directions of change: a strong increase in the number of thermophilous species was observed in 5 on 7 transects (Fig. 3) and was accompanied by the parallel reduction in microthermic elements. For instance, *Pieris callidice*, a microthermic, xerophilous species of the open herbaceous environments of the alpine belt, disappeared completely.

We also observed a strong general ingression of the widespread and common species (e.g. Vanessines), all characterised by high vagility and high ecological tolerance, while little mobile species came in reduced frequencies. Species linked to the wooded habitats (nemoral species) strongly increased in frequency, apparently to the expenses of species linked to the open herbaceous environments occurring above the tree-line (subalpine and alpine belts).
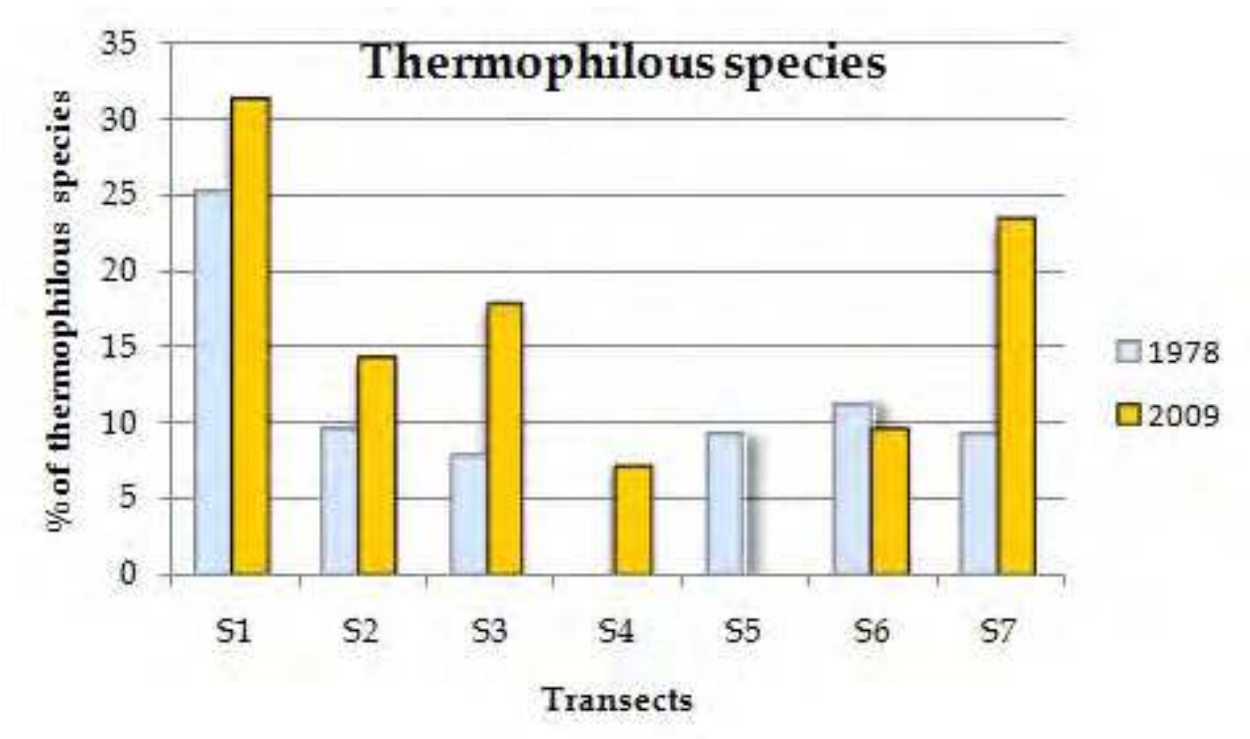
Fig. 3. Percentage of thermophilous species in each transect (classified as S1-S7) through the analysed time frame. Different colours refer to different sampling years (blue=1978; yellow=2009).

The sampling design adopted for the 2009 field season allowed us to demonstrate a phenological shift in the flight period of some species, which in 1978 occurred between the end of July and the beginning of August, but which we only recorded at the beginning of the sampling season, in 2009. The case of *Anthocharis cardamines* is somewhat peculiar, since in 2009 it reached the peak of its flight period in the middle of June and was never observed after the beginning of July, while in 1978 it was observed in 5 of 7 transects at the beginning of August.

Changes observed in community compositions and phenologies apparently went in the direction of the predicted impacts of climate changes. We did not observe any decrease in species richness, but rather a number of changes in the composition of species assemblages. Indeed, species responded individually to the changing environmental conditions (see Hengeveld, 1990). In particular, we wish to stress the loss or strong decline in a number of species whose characteristics made them potentially sensitive to climate change, e.g. the geographically localized, the poor dispersers, or the ecologically highly specialized (see McNeely, 1990).

The analysis of habitat changes identified by photo-interpretation revealed strong variation in land cover between 1978 and 2006 (most recent pictures available), with a generalised spread of forested areas (trees and/or shrubs) and a reduction in the size of grasslands. This is fully in agreement with changes observed in the composition of butterfly communities as regards individual species and their environmental preferences. The observed increase in vagile and tolerant species perfectly fits to the same pattern. The reduction of open surfaces, in fact, will not be a problem for the highly mobile species, which can easily reach another suitable site, while the less mobile species will remain 'locked' in their continuously

shrinking micro-environment (Boggs & Murphy, 1997; Shreeve, 1995). In the same way, broadly tolerant species will take advantage of a '*changing environment*', because of their strong adaptability, to the detriment of the stenotopic and more specialized species (Krauss et al., 2003; Shreeve, 1995).

In contrast, a first analysis of our climatic dataset did not provide any strongly supporting information. The analysis of temperature data did not show any clear pattern. Within the time frame between the beginning of the 1980s and 2010, we remarked only a slight increase in maximum temperature. A similarly unclear pattern was recorded by the analysis of precipitations, both in term of rainfall and in number of rainy days. Even though climatic data will have to be analysed more in detail, it is difficult, at the moment, to understand which factor(s) may be responsible for the observed vegetational shifts, or may have determined the observed changes in butterfly communities.

As we have already pointed out, the influence of climate change in the European Alps is regionally confused by human activities. Cattle grazing in the alpine pastures has been decreasing throughout the last century, allowing a fast recolonization by trees and shrubs, everywhere the treeline had been artificially lowered (Vittoz et al., 2008). Most of the observed changes in butterfly communities, however, cannot be explained only in term of changes in land use: the general increase of thermophilous species and the reduction or disappearance of some alpine and microthermic species can be a signal of a changing climate.

The same is to be said for the observed phenological shifts, even though the latter might be a consequence of weather variability during a single season, rather than of a more general pattern of warming climate.

The importance of this work derives from the fact that many previous studies aimed at understanding the effects of climate changes on community composition, relied on the comparison of contemporary data with historical data sets (atlases, collection specimens), which were often collected in a non-standardized way and referred to a much coarser spatial grain. This makes difficult to exactly detect altitudinal changes in species distribution, as well as to analyse the respective roles of single environmental factors. For these reasons, transects set in well-specified areas are more appropriate for investigating the altitudinal range shifts occurring in response to contemporary climate warming (Archaux, 2004). We suggest, however, that to gain full understanding of the underlying patterns and the relative roles of different causes, it will be fundamental that transects are repeated both in space (i.e. along altitudinal gradients) and in time, over a suitable number of years. Our data from the W Italian Alps, taken on exactly the same sites at a 30 years distance fulfil, in principle, all these needs.

2.3 The ingression of alien species

Climate change and global warming have promoted, in the last decades, the arrival and expansion of several alien species in Europe. Over the past 60 years it is estimated than about 130 exotic pests have become acclimatized in Italy and 7% of them are Lepidoptera. Most aliens species came from America, Asia, Africa and Australia (Pellizzari et. al., 2005).

The main risk factor for alien species entry is the accidental introduction owing to intensive commercial exchanges of plants and goods and an ever-increasing tourist traffic. For millennia, the natural barriers of oceans, mountains, rivers and deserts provided the isolation essential for unique species and ecosystems to evolve. In just a few hundred years

these barriers have been rendered ineffective by major global forces that combined to help alien species travel vast distances to new habitats and become invasive species.

The impacts of alien invasive species are immense, insidious, and usually irreversible. They may be as damaging to native species and ecosystems on a global scale as the loss and degradation of habitats. The scope and cost of biological alien invasions is global and enormous, in both ecological and economic terms, indeed the direct economic costs of alien invasive species run into many billions of dollars, annually.

Cameraria ohridella is a tiny moth of the family Gracillariidae and a very well known leaf-miner of the common Horse Chestnut. It was first observed, and immediately described as a new species, in the northern Greek region of Macedonia, where its food-plant is native, during the early 1980s, but it quickly invaded most of Europe (Austria, Italy, France etc) over the following two decades. The rapid geographical expansion of this parasite was mainly correlated to the passive transportation of infested plant leaves or adult moths, probably in cars or in other vehicles. As a consequence, it soon reached most of the big cities of Central and South Europe, where the Horse Chestnut is abundantly cultivated as an ornamental tree (Ferracini & Alma, 2008). Larvae develop almost exclusively on the white-flowered (not on the red flowered) species (*Aesculus hippocastanum*) to which they cause significant damage by digging their way into the leaves and stunting growth. Infected leaves are covered in small brown patches, which rapidly spread across the entire tree. They give trees an autumnal appearance and cause important aesthetic impact. The invasive success of this parasite relates both to its high rate of population growth and high dispersal capability. Once established in a site, its populations reach outbreak densities within a few years. Several generations of this moth develop per year whilst the impact of natural enemies is still limited (Settele et al., 2010). The low incidence of parasitism of the horse chestnut leaf-miner by European parasitoids, in fact, is at least partially a matter of poor synchronization between the life cycles of the native antagonists (parasitoids) with that of the introduced host moth. Therefore, the lack of suitable host when the parasitoids are ovipositing will inhibit their adaptation process (Grabenweger, 2004).

Another moth, *Paysandisia archon,* of the Castniids family, is native to Central America (Argentina and Uruguay) and was accidentally introduced into Europe, where it is spreading rapidly. In its natural range, the species is not considered a pest, but in Europe it is causing increasing concern because of the sometimes fatal damage produced to both native and exotic palm trees. The main symptoms of this infestation are in the deformation of leaves, some of which become haggard, yellow in color and bored. Larvae dig tunnels through the stem or the young leaves, causing characteristic damage (Vassiliou et. al., 2009).

Yet another, slightly different, example is provided by *Danaus chrysippus*, a widespread species in the paleotropics, which was observed for the first time in Italy (Campania) by Ochsenheimer in 1806. It soon became extinct in that area, only to reappear in Apulia in 1983. While there are doubts on whether or not its first arrival was due to passive introduction (contemporary authors blamed ships of the British Navy), its reappearance was quite natural, and probably a consequence of global warming. During the last 20 or 30 years the species has spread to the whole Mediterranean basin. Its larvae grow on Asclepiadaceae originally imported for the production of 'wild cotton'. Strictly speaking, therefore, we cannot consider this a really alien invasive species. The apparently recent arrivals of *Zizeeria knysna* (see Romano & Romano, 1995) to the tiny Italian island of Lampedusa, the recent burst of invasiveness shown by *Thaumatopoea pityocampa*, or even the fluctuating invasiveness of *Lymantria dispar* are to be viewed in a similar framework.

Some studies have highlighted that only a tiny proportion of non-indigenous species become invasive and most invasions occur in man-dominated rather than in pristine ecosystems. Moreover, indigenous and non-indigenous species are sufficiently similar that their impacts may not necessarily be different and there is evidence that introduced species will sometimes augment, rather than reduce, species diversity (see Hulme, 2003 for a detailed analysis).

However, we believe that invasive alien species are a major threat to biodiversity, as well as, sometimes, livelihoods. Many countries have limited resources to prevent the introduction and spread of invasive species, or to prioritise and implement surveillance, eradication and control. A key constraint is in the lack of direct access to suitable data and information. On a global scale, data on invasive species are sparse, geographically biased, of variable quality, and expensive to obtain. Yet, the mobilisation and improved accessibility of existing global information would represent a significant resource at national and regional levels.

Different kinds of information are needed for assessing risks of establishment, spread, negative impact and difficulties in management. For each species, information requirements for spatial modelling include native and introduced range, point occurrence/observation data as well as climate layers at appropriate resolutions. Another key piece of information is 'invasiveness elsewhere', since "only one factor has a consistently high correlation with invasiveness: whether or not the species is invasive elsewhere" (Wittenberg & Cock, 2001).

Generally speaking, hundreds of extinctions may have been caused by alien invasives. The ecological cost is in the irretrievable loss of native species and ecosystems, while the direct economic costs of alien invasive species may annually run into the billions of dollars. It is in this framework that the relevant IUCN SSG has identified the problem of alien invasive species as one of its major initiatives at a global level (see http://www.issg.org/database/welcome/).

2.3.1 Study case: The Geranium Bronze

The butterfly *Cacyreus marshalli* (Lepidoptera: Lycaenidae), commonly known as Geranium Bronze (Fig. 4, a-c), is an invasive species native of South Africa currently occurring in many parts of Europe and the Mediterranean area.

During our studies we have investigated the *C. marshalli*'s ability to spread to native *Geranium* spp. and evaluated the conservation risks that such a shift would pose for both native geraniums and cohabitant butterflies. In Europe, *C. marshalli* larvae normally feed on pelargoniums causing damage to flowers, stems and leaves (Fig. 4, b). Seriously affected plants may die as a result of the infestation. As a consequence of the fast spread and dangerousness of this pest, pelargonium were included among the species requiring quarantine and *C. marshalli* was listed as a A2 quarantine pest by the European and Mediterranean Plant Protection Organization. The key factor of its invasiveness is the relationship with the host food plant: in particular its potential to spread to native *Geranium* spp. Recently the host plant preferences of the Geranium Bronze were investigated under controlled conditions (Quacchia et al., 2008). Studies included 9 Italian native *Geranium* spp., which commonly occur in many mountainous and hilly habitats of north-western Italy, showed that under no choice conditions, at least one egg was recorded on each tested plant, except for *G. phaeum* and all the plants on which oviposition occurred were fully suitable for larval development. No statistical differences were detected in the wingspan between adults emerged from *Geranium* (Fig. 4, d) and *Pelargonium*. *Cacyreus marshalli* represents a

potential threat for both native geraniums and for geranium-consuming lycaenids, such as *Aricia nicias* and *Eumedonia eumedon*. Since the Geranium Bronze is multivoltine and accepts to lay on leaves and stems and not only on flowers, it would probably out-compete the other two, both monovoltine, in areas of sympatry. As concerns the probability that a shift onto the native geranium species can occur in natural conditions, the voltinism-suitability hypothesis suggests that the diet breadth is restricted to hosts supporting the most rapid larval development where populations are near the thermal limits. In contrast, when the same number of generation can be achieved without difficulty, relaxed selection will permit females to oviposit on hosts on which larval growth is lower (Scriber & Lederhouse, 1992). Variation in oviposition preference behaviour among different host species is heritable and responsive to selection (Renwick & Chew, 1994; Thompson & Pellmyr, 1991). The evolutionary mechanisms allowing or favouring the shift to a "new" larval host plant are rather obscure. Some authors have argued that species invasions can increase, rather than reduce, species diversity (see Hulme, 2003 for a detailed analysis). This may be true also for *C. marshalli*, but only in man-dominated, urban ecosystems. Should this species be able to spread to natural environments it would probably cause considerable ecosystem impact, by affecting both native geraniums and geranium eating lycaenids. Even apart from this, as suggested by Trematerra & Parenzan (2003), the adaptation of this lycaenid to autochthonous plants may favour its spread and establishment not only in Italy and in Mediterranean areas, but also deep into mainland continental Europe, causing serious economic and environmental losses.
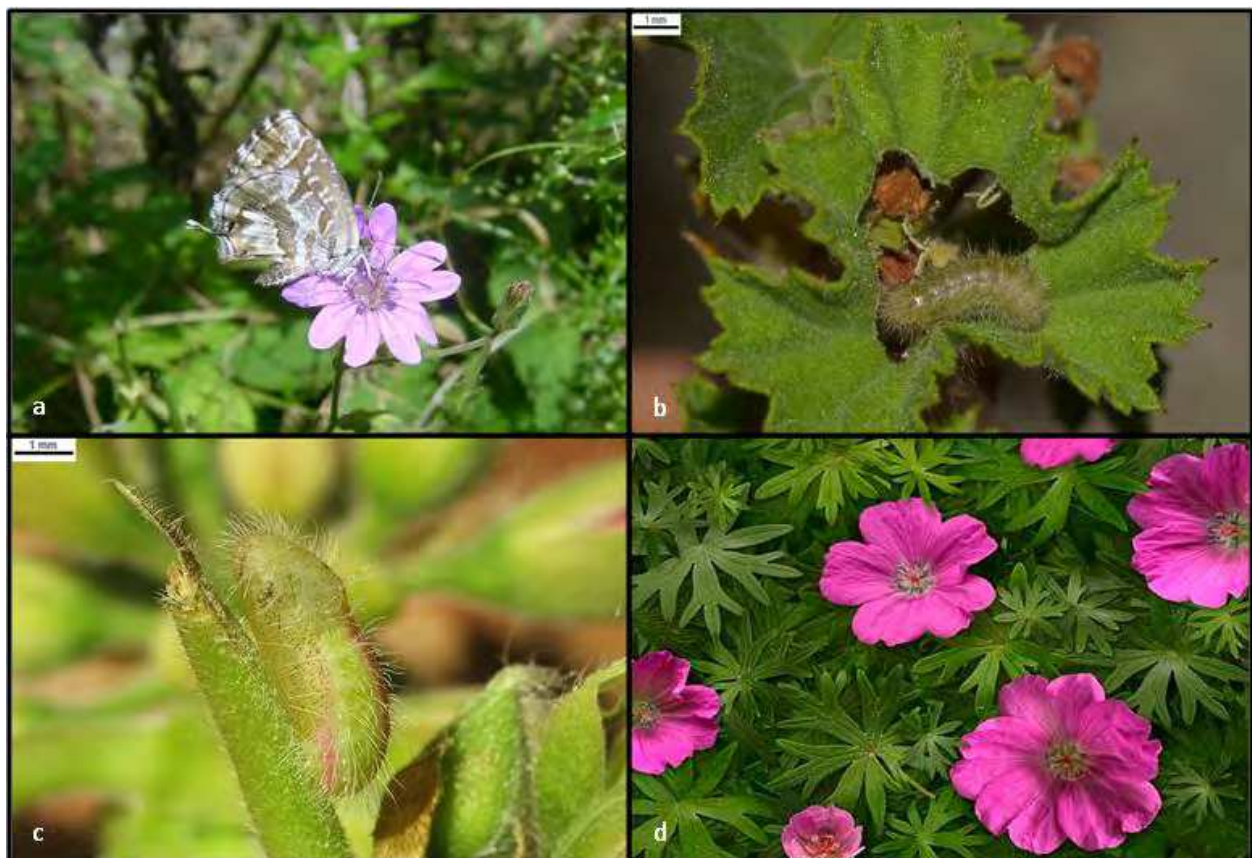

Fig. 4. *Cacyreus marshalli:* a) adult; b) larva; c) pupa; d) *Geranium sanguineum.* Courtesy of S. Canterino and D.S. Ossino.

Cacyreus marshalli has never been reported as a pest species in its area of origin, probably because some autochthonous parasitoids and predators are able to keep its population under the damage threshold. In our study we have not observed any parasitoid or predator. Among parasitoids, only Sarto i Monteys (1992) reports one egg of *C. marshalli* parasitized by *Trichogramma evanescens* Westwood (Favilli & Manganelli, 2006). The introduction into Europe of the Geranium Bronze is having great impact on the plant-nursery sector, with a consequent decrease in the demand of *Pelargonium*, which are ever more often replaced by customers with other ornamental plants. Even though in nurseries the control of the pest may be carried out through the aid of common insecticides, this is not feasible elsewhere, particularly in the mountainous and hilly habitats where wild Geranium species (Fig. 4, d) commonly occur. For the regions of Piedmont and Aosta Valley, Pignatti (2002) reports the occurrence of 18 Geranium species. Should an adaptation to these native species ever occur, *C. marshalli* would become a real threat for the native flora, as well as for local biodiversity.

3. Conservation

Biological conservation is, by definition, a work in progress. Biodiversity is under ever increasing or sometimes shifting threats, invariably, although sometimes indirectly, generated by human intervention. The task of preserving biodiversity is increasingly difficult and none of our results, no matter how apparently positive in the short or intermediate term, is to be taken as granted forever. Humans increase in numbers, and even more so is increasing their use of land and energy; climate, at least partially as a consequence, changes to unprecedented rates; vegetation structure becomes disrupted and landscapes change, becoming increasingly uniform and banal. A conservationist's work is never done.

This does not necessarily mean, however, that biological conservation is a doomed initiative, which will never see any lasting success. Many new instruments of intervention have become available to conservationists in the past several years, and many new ones may be expected to be developed in the near future.

We will discuss some of these new instruments in the following pages.

3.1 The Italian biodiversity hotspots

As it has been mentioned already, Italy is a highly species-rich region, in comparison to central and northern European countries, and the Italian fauna is generally well known, at least taxonomically and geographically. A minimum of 59, 302 native Italian animal species are currently known to exist, not less than 5845 of which are endemic to the Italian political territory.

For N European scientists, travelling in such a biodiversity-rich region may certainly be a dazzling experience, but having to deal with such a complex fauna makes a conservationist's work extraordinarily difficult. Species interact in many ways, they form very complex food-webs and they co-operate or compete with each other in unexpectedly intricate ways. Most of these interactions are still unknown, many are only suspected or insufficiently substantiated. The embarrassing truth is that for most of our extraordinarily rich fauna we only have descriptions of the external morphology, sometimes, but not always, accompanied by some detailed information on their geographical distribution over the Italian territory. Even this, however, may be important from a conservationist's point of view.

Identifying regions having extraordinarily high species richness has been recognised as an important conservation tool. The notion that some areas have highest biological diversity (biodiversity hotspots) has been long established. Originally Myers et al. (2000) defined biodiversity hotspots as areas "where exceptional concentrations of [endemic] species are undergoing exceptional loss of habitat", but this concept has become increasingly generalised, in more recent times, to cover all aspects of biological diversity. Hotspots of species richness are generally large areas, where conservationists will be able to create one or more reserves covering a broad range of species. A shift of emphasis from the macro-geographical to the local scale, however, has generated several difficulties. At the regional level, hotspots may harbour large numbers of common and widespread species, rather than endemic, rare, or threatened species. In other cases, they may represent areas where field studies, rather than species, have been concentrated and species distribution reflects the distribution of taxonomist, or of their sampling efforts.

In a recent work (Balletto et al., 2010) we analysed species composition across the whole Italian political territory and over a relatively broad sample of taxa, each characterized by a range of often idiosyncratic ecological requirements.

The presence or absence of individual species was sampled over a 10x10 km UTM grid and across the Italian territory, considering 471 species of zygaenids, butterflies, carabine coleoptera, amphibians and reptiles. We analysed their distributions in terms of hotspots of species richness, of rarity and of complementarity and compared the efficiency of these methods in evaluating local animal diversity. Our aim was to compare methods to assess biological diversity, and we were not dealing, as a consequence, with functional biodiversity.

We carried out this work both at national and at regional level, trying to understand if high diversity areas extracted at national level could be taken as representative of regional situations. At national level, out of 3218 10x10km UTM quadrats sampled, 161 (5% of total) had highest species richness. Islands included only 1 hotspot (Sicily). Sixty-eight species (14.4%) were not represented. They were mainly endemic (65%), insular (73.5%), or rare (25%). Working taxon by taxon, hotspots increased to 433. Only 85 (19.6%) were hotspots contemporaneously for two taxa and only 9 were hotspots for 3 taxa. Missing species were fundamentally insular species. The regional-level approach generated 467 hotspots. Eleven species were not represented (2.3%). They had marginal distributions, or were insular endemites. Hotspots of rarity numbered 235 and 10 species were not included. Results demonstrated that hotspots of high species richness are poor predictors of overall biodiversity and that they rarely coincide among animal groups.

The complementarity method identified 67 quadrats (Fig. 5). By definition, complementarity quadrats accounted for all species investigated. They failed, however, to predict the occurrence of three zygaenids, which were previously not included in the analysis. As expected, complementarity provided better results than hotspots analysis and one can assume that combining the two methods will assure that areas having the highest biodiversity values are identified. Regional or rarity hotspots should generally be preferred to hotspots of species richness and among the latter those calculated at regional level should generally be preferred, as well as hotspots of rarity. At the moment, the perfect method for assessing regional biodiversity probably does not exist. Whereas hotspots analysis aims at providing an absolute measure of local biodiversity, complementarity does not and takes a more pragmatic approach to intermediate scale conservation practice. Combining the two
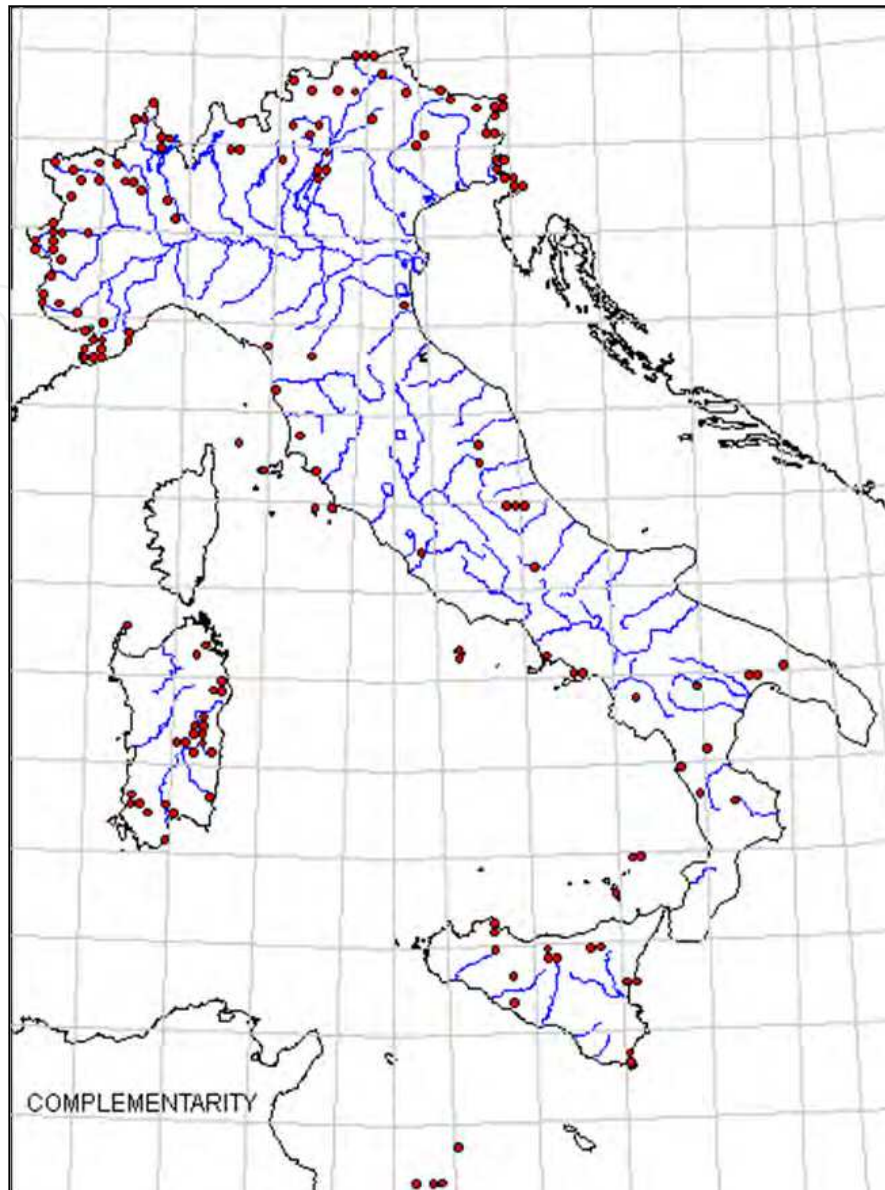
Fig. 5. Quadrates identified by complementarity analysis.

methods will assure that areas having the regionally highest biodiversity values are identified, even while working with necessarily incomplete or skewed databases. In these areas, conservationists will be able to create one or more reserves covering the broadest possible range of species, for that particular region.

3.2 The NATURA-2000 network

Partially shifting the emphasis from the community to the species level, the European legislation and in particular the "Birds Directive" (79/409/EEC) and the "Habitats Directive" (338/97/EEC) respectively require Member States to designate "special protection areas" and "special areas of conservation" that will eventually become integrated into the NATURA-2000 network of Sites of Communitarian Importance (SCIs) and finally coalesce into Special Areas of Conservation. Sites to be included within this network are designated either to ensure the appropriate conservation of some [bird or otherwise] species

deemed to be threatened within the European territory, or of a combination of other threatened animal or plant species, and of their habitats.

From a conservationist's perspective, once the designation phase is completed, this approach is expected to ensure that all species and habitats of Communitarian Interest may be afforded suitable protection. As an indirect consequence, however, it is possible that many other species, either not threatened or unknown to be threatened, are also potentially protected, in a way that is in many respects similar to the "umbrella" concept of some Conservationists.

The overall effectiveness of this process in generating a coherent network of globally important conservation areas, however, has been tested rarely and effects of its implementation on a local basis have been questioned (Troumbis & Dimitrakopoulos, 1998; Dimitrakopoulos et al., 2004).

As concerns the Italian NATURA-2000 network in particular, Maiorano et al. (2007) have investigated its potential effectiveness in affording terrestrial vertebrate species a favourable conservation status. For their analysis, these authors used available distribution models (DMs). These models, however, are generally not available for invertebrates, which represent a conspicuous percentage of threatened species.

Recently (Bonelli et al., [submitted]) we tried to assess, by actual distribution data, the proportion of the Italian terrestrial invertebrate and small vertebrate species that will be included in protected territory once the NATURA-2000 network is set in place.

For each of 429 species investigated, we calculated the ratio between the number of data of presence within the SCIs/SPAs protected sites and in non-NATURA-2000 areas ("observed value"), and we compared this value with the ratio between the areas of protected and non-protected sites ("expected value"). For individual species, the "expected value" represented the proportion of presence data that are expected to fall within a NATURA-2000 area, in case presence data were scattered at random.

Most species (427 of 429 analysed) were represented at the sites proposed for the Italian NATURA-2000 network. Only 2 species were missing, namely one butterfly (*Boloria eunomia*) and one carabid species (*Carabus* (*Hygrocarabus*) *nodulosus*). They are both rare species, which marginally enter the Italian territory, where they occupy respectively 1 and 2 UTM quadrats. Neither of them is considered threatened in Europe, at least by the Habitat Directive.

The Italian NATURA-2000 network (Fig. 6a), once fully implemented, will represent a suitable instrument to foster the conservation of many invertebrate and smaller vertebrate species. As we said, since a vast number of these sites are coincident with areas encapsulating very high biodiversity (Balletto et al., 2010), their general importance may partially transcend the currently prevailing species-by-species approach to biological conservation. The geographical distribution of SCIs on the Italian territory ensures that they include a significantly larger number of species than random. Most of the Italian NATURA-2000 sites, however, are small and this, of course, does not make them very suitable for some larger vertebrates (Fig. 6b). As it is the case in most of Europe (Gaston et al. 2008), in fact, the geography of the Italian territory, combined with high human-population densities makes identifying more numerous large-sized NATURA-2000 areas impossible. Large NATURA-2000 sites are often located within the "historical" Parks, which were created well before European Directives were even conceived, and at a time when the "ecosystem level" conservation philosophy was prevalent, at least in Italy.

In this work we showed that in most cases the main problem is the lack of type of management on which species' survival depends.

As we have already shown, a further threat to Italian biodiversity will be represented by climate change. Range shifts will pose major problems to conservation, as species that are already protected might actually shift their distribution out of the currently protected areas which were designated to preserve them. Networks of designated sites, including the NATURA-2000 network, should be managed flexibly.
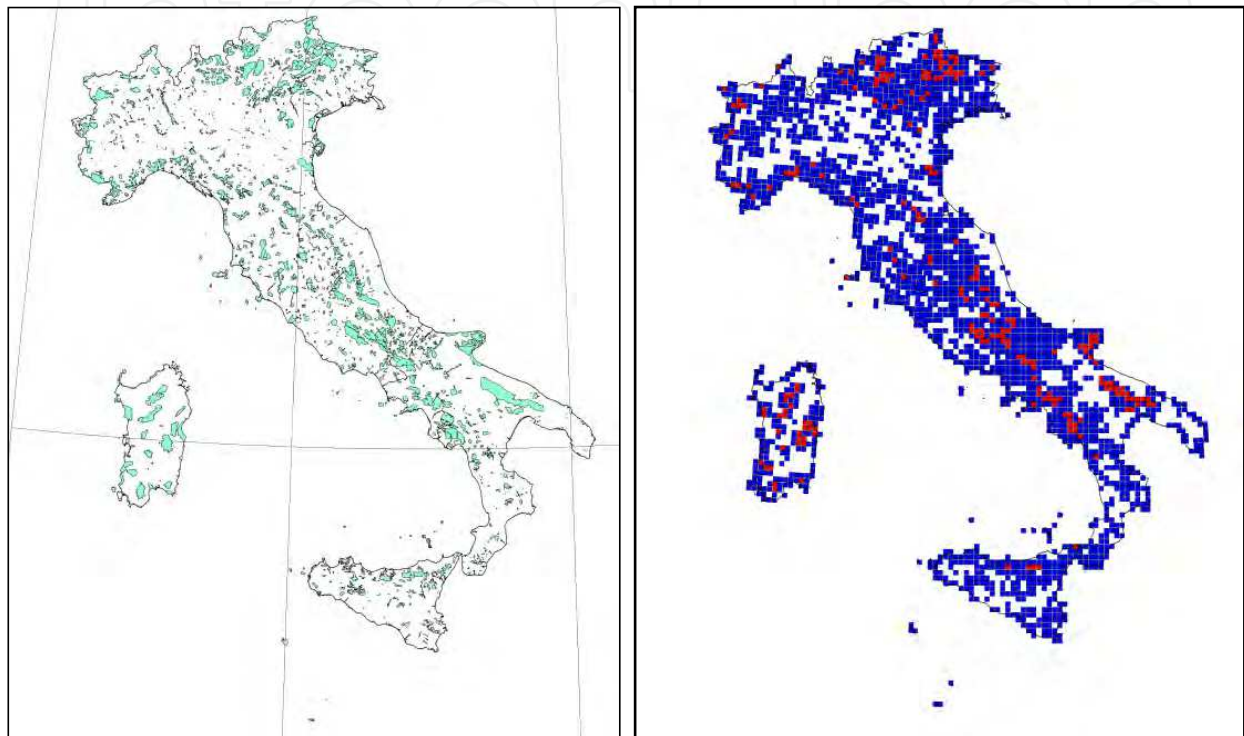

Fig. 6. a) The Italian NATURA-2000 network of protected sites (terrestrial sites only); b) Italian UTM-MGRS quadrats including over 10 ha (blue) or over 5000 ha (red) of NATURA-2000 protected sites.

3.3 The European "Red list"

The European Red List is a review of the conservation status of about 6,000 European species of dragonflies, butterflies, freshwater fishes, reptiles, amphibians and mammals, as well as of some selected groups of beetles, molluscs, and vascular plants. Ranks, ranging from Extinct (EX) down to Near Threatened (NT), or Data Deficient (DD), Not Evaluated (NE), etc. have been given to individual species according to the IUCN (International Union for Conservation of Nature) regional Red Listing guidelines (IUCN 2001). In 2010, the IUCN, together with Butterfly Conservation Europe (BCE) and in collaboration with the European Union published the Butterfly Red Data Book (van Swaay et al., 2010). The conservation status of a total of 482 European species was assessed for the purposes of this list. Almost a third of these species (142) are endemic to Europe. Species classified as either threatened (i.e. Vulnerable (VU), Endangered (EN), or Critically Endangered (CE)), or Near Threatened (NT) represent 19% of the total and are thus deemed to be of high conservation priority. A high proportion of threatened and Near Threatened butterfly species are endemic to either

Europe or the EU. This is particularly true for France, Italy, Spain, Greece and Bulgaria and, more in general, most of the threatened species are confined to one or more parts of southern Europe. The main current threat to European butterflies was recognised to be in the loss of their habitat. In many cases habitat connectivity is also important, being challenged by changes in agricultural practices, either through intensification or abandonment. Other important threats are climate change, increased frequency and intensity of fires and development of tourism and related infrastructures. As BCE partners, we provided all the necessary data on the Italian butterflies. Based on this list, in Italy we have three Endangered (EN) butterflies (*Maculinea arion*, *Polyommatus humedase* and *Coenonympha oedippus*), as well as seven Vulnerable (VU) species, while another 21 species are deemed Near Threatened (NT).

3.4 The "Butterflies Prime Areas"

The principal aim of the "Prime Butterfly Areas" European project was to identify the most important areas where conservation efforts should be focused as a matter of urgency. The objective of this project was of implementing the Red Data Book of European Butterflies and to suggest management measures for these areas. A site was classified as a Prime Butterfly Area if it contained a substantial resident population of at least one species which is threatened according to the Red Data Book of European Butterflies (van Swaay & Warren, 1999), or to the IUCN Red List of threatened animals. These species are called target-species. As with the Red Data Book, all data were provided by national compilers who were asked to select the 'best', in this respect, areas for every country, and to accompany their lists with all necessary information on location, key butterfly species, land uses, threats, and needed or already implemented conservation measures.

Following the criteria set by van Swaay & Warren, 32 Prime Butterfly Areas (PBA) have been identified in Italy as a first selection of the most important areas for 16 species deemed to need protection (Balletto et al. 2003 [in van Swaay et al. 2003]).

The main purpose here is therefore to inform about this selection and to provide definitions for the sites forming the Italian section of the NATURA 2000 Network. Italian Prime Butterfly Areas were identified by using 16 target species. In total, they cover 365, 209 ha of Italian territory.

The 16 target species used to identify the Italian PBA are: *Argynnis elisa, Coenonynpha oedippus, Erebia calcaria, Erebia christi, Euphydryas aurinia, Lopinga achine, Maculinea arion, Maculinea rebeli, Maculinea teleius, Melanargia arge, Papilio hospiton, Parnassius apollo, Plebejus trappi, Polyommatus galloi, Polyommatus humedase, Pyrgus cirsii*.

Two of them, *Polyommatus galloi* and *Polyommatus humedase* (IUCN global threat status: Endangered, based on the Italian Red List) are restricted to as many small areas and represent 100% of the global (and of course European) population of those species.

Rare and threatened butterflies are not widely distributed species and often occupy very small sites. In the 32 Italian PBAs the most common types of land-use are nature reserves, touristic and agricultural areas.

As concerns these areas, a very important threat is in their isolation, since the distance between two areas is sometimes too large to allow adults' interchange. Also management types (or lack of management) and hectic natural events represent a common problem. All sites are in strong need that active management practices are planned and implemented as soon as possible. Providing a correct management of Prime Butterfly Areas will not only

support the conservation of a few target-species, but will allow to reach a number of other far-reaching objectives fostering the conservation of many butterflies and insect species in general.

Preserving biological diversity for the future generations is the single most important objective of conservation biology.

3.5 The role of the Alps in butterfly diversity conservation

Many European butterflies species are restricted to the alpine, sub-alpine and mountain habitats with 25 species confined in the Alps and other 66 with a wider distribution in mountain habitats. The Alps are among the most important hotspots of endemism in Europe (Essl et al., 2009). As concerns their distributions, narrow-range species were strongly influenced by Pleistocene glaciations, both in the case of plants (Tribsch, 2004) and of some invertebrates, including butterflies (Hüemer, 1998). Since the Alps are the largest and highest mountain system in Europe, it is not surprising that a high number of endemic species, typical of the mountain environments, is found here, as observed in many Lepidoptera and especially in micro-moths (Varga & Schmitt, 2008). Some of these endemic species are distributed throughout the Alps, while others are local endemism found only in parts of them. Two types of local hotspots of endemism have been identified. The first group includes species whose ranges are restricted to peripheral regions of the Alps and mainly confined to the intermediate and low altitudes, as observed in some species of the genus *Erebia* and in some Lyceanids (Varga & Schmitt, 2008). The highest concentrations of these local endemics are found in the southwestern and southeastern regions of the Alps, which include large areas at low altitude, not covered by ice during glaciations. These areas were probably used as centers for glacial survival, where these species experienced shifts in altitude but not expansion of their range on a large scale (Schmitt, 2009). Shifts in elevation, in response to Pleistocene climatic fluctuations, may have repeatedly promoted evolutionary processes and speciation. The second group is represented by species confined to the inner parts of the Alps, such as a large number of micro-moths (Hüemer, 1998) and some butterfly species (i.e. *Erebia nivalis*) (Varga & Schmitt, 2008). These species are generally restricted to high mountain habitats and might have survived glaciations in small ice-free areas, on the slopes of the inner Alpine mountains (Schmitt, 2009).

The Italian fauna includes 283 species of native butterflies (Balletto et al., 2007), 106 of which are located in the Italian Alps (25 are strictly alpine) and 64 in other mountain habitats. A recent study has identified the Alps as one of the most important biodiversity hot-spots of the Italian peninsula combining the use of the niche- modeling with a complementarity-based method called zonation (Giradello et al., 2009). In the Italian Alps, butterfly species form loose associations, characterized by low inter-specific competition for space and other resources (Tontini et al., 2003). Density-independent processes (Den Boer, 1998) generally determine the population sizes. Butterfly communities are composed by various combinations of stenotopic species, occurring together with some erychorous or, sometimes, migratory species. Most of the butterfly associations are inseparably associated with vegetational types, along a successional gradient. This seral connection, however, breaks down above the treeline, dominated by grasslands where the ecological conditions are characterized by a unique set of physical and biotic parameters (Nagy, 1998; Ozenda, 1985). These grasslands are well represented in the Italian Alps, and are generally less affected by human activities than at lower elevations (Tontini et al., 2003).

Perhaps the single most important factor that will affect the mountains system in the future is global warming, which will also have severe effects on animal communities. In the last two decades, climate changes were associated with increases in northern latitudinal limits (Hill et al., 2002; Parmesan et al. 1999), elevational shifts (Descimon et al., 2006), contraction of range size (Parmesan et al., 1999) and phenological modifications (Roy & Sparks, 2000; Stefanescu et al., 2003). At high altitudes, continuous changes in temperature and other climatic parameters are expected to have a strong effect on plants and the associated animal assemblages (Beniston et al., 1996; Walther, 2003). A study on *Boloria titania* and its foodplant *Polygonium bistorta* strongly suggests that climate change has a potential to disrupt trophic interactions because co-occurring species do not react in a similar manner to global change. *B. titania* might considerably expand its future range in case its host plant (*Polygonum bistorta*) has unlimited dispersal, but it may lose from 50 to 75% of the its current distribution if the host plant will be unable to fill its projected ecological niche space, or from 79 to 88% if the butterfly is assumed to have highly limited dispersal (Schweiger et al., 2008).

Climate change is already impacting many populations across Europe (in particular the tundra species, such as *Colias hecla* and *Euphydryas iduna*) and is likely to affect additional species even more significantly, in the future (Settele et al., 2008). In the recent Atlas of Climate Risk, Settele et al. (2008) examined the current distributions of 294 European butterflies and evaluated their future ranges at the years 2050 and 2080, on the basis of variations projected on 22 climatic variables. Their results predict important losses in climatically suitable areas, particularly in the South of Europe and the Italian peninsula. The Alps, in contrast, where many SCIs and Bird areas are located, will further increase their importance for biological conservation, since many stenotopic species will become concentrated in this area. Strict stenotopy is generally a consequence of adult behavior, rather than the larval biology and the genus *Maculinea* offers a rare example in Europe.

3.5.1 Habitat preferences of *Maculinea arion* in the Alps

Projections of climate change scenarios predict that many of *Maculinea arion* populations will disappear from Europe in the next 50 years and those close to the southern limits of the species' range will be the most threatened (Casacci et al., 2011). While the vast majority of the Italian populations of *M. arion* are expected to become extinct, some will survive in mountain areas, especially in the Alps (Settele et al., 2008 – Fig. 7).

M. arion is a lycaenid parasite of ants of the genus *Myrmica* and its survival depends on the presence and abundance of two resources, a specific foodplant, *Thymus pulegioides* and one or more *Myrmica* host ants. We collected data on the distribution of thyme and on the abundance of *Myrmica* ants. We measured sward height close to the each nest, as an indicator of the microclimatic niche, the distance between the ant colonies and the butterfly foodplant, as well as the structure of vegetation patches in 14 in the Western Italian Alps (Val Ferret: Aosta) (Casacci et al., 2011).

Results from our study showed that colonies of different *Myrmica* species occur at different distances from *T. pulegioides* plants, probably as a consequence of their different microhabitat requirements. *M. sulcinodis* (one of the host-ant species recorded together with *M. lonae*) colonies occur in the nearest range with respect to *M. arion* foodplants, which can suggests that they have similar habitat demands as those of Thymus foodplant. Therefore, larger overlapping and smaller distances can results in higher probability of adoption for *M. arion* larvae, since a higher number of foraging workers can be found in the near

Fig. 7. Alpine habitat of *Maculinea arion.*

surrounding of Thymus plants. *M. sulcinodis* is, moreover, also the most abundant *Myrmica* species on pastures with a high number of *M. arion* adults, whereas it is a scarce species on patches with lower butterfly abundance. It seems, therefore, that *M. arion* mostly inhabits patches where these two resources fully overlap each other.

Thomas and his colleagues (1998b) have pointed out that, for conservational purposes, it is necessary to have precise information about the sward height and the niche preferences of the host ant species. In England, a strong correlation between the turf height and the *Myrmica* community was found (Thomas et al., 1998b, 2009). In particular, M. *sabuleti* (the unique host-ant of English *M. arion* populations) has a very narrow niche, corresponding to a grass height ranging between 0 and 3 cm. The ant's density decreases with increasing grass height, until disappearing over 7 cm (Thomas et al., 2009). Thus, inconspicuous changes in grazing regime and vegetation structure have caused the host ants to be replaced by similar but unsuitable congeners, explaining the extinction of *Maculinea arion* populations in 1979 (Thomas et al., 2009).

In Val Ferret, the host specificity of *M. arion* is apparently not limited to one single species of the genus *Myrmica*. Moreover, it appears that each *Myrmica* species requires relatively broader niches in relation to the grass height. In the Italian Alps, the greater tolerance of the *Myrmica* species with respect to increasing grass height, may be the result of much more pronounced daily and monthly temperature fluctuations. Data obtained in Val Ferret during July 2009 show that the average daily maximum temperature was 27 °C,

while the minimum temperature was about 11 °C. In contrast, climatic data collected in England at *M. arion* sites show that the average daily maximum temperature is 19.5 °C while minimum temperature is 13.5 °C. Thus, the much more pronounced daily and monthly fluctuations occurring in Val Ferret might have pushed *Myrmica* species to become much more adapted to broader niches and tolerant with respect to marked changes in the sward height.

3.5.2 Larval ecology of *Colias palaeno*

Colias palaeno (Lepidoptera: Pieridae) is classified as vulnerable to climate change by the authors of the '*Climatic Risk Atlas of European Butterflies*' (Settele et al., 2008). In recent years it has undergone a great decline in many parts of Europe, in particular in the low altitudes of South Germany, where it is present only on some peat bog margins, while it disappeared in the last ten years from many, apparently unchanged, habitats (Dolek et al., 2007). One possible explanation could be linked to larval survival, before and/or after overwintering. Peat bogs are environments characterized by high humidity levels on oligotrophic soils, and are potentially vulnerable to changes in air temperature, rainfalls regime and general atmospheric composition. These changes can determine micro-habitat alterations, including micro-climate and larval host plant characteristics, and consequently the performance and feeding behaviour of the small larvae. *Colias palaeno*'s single larval host plant is *Vaccinium uliginosum*. In the lowlands of South Germany this plant is only present in peat bogs. In the Alps, in contrast, it grows in many different ecological conditions, even in much drier places, such as the *Vaccinium* and *Rhododendron* mountain heaths. In the Alps, *C. palaeno* populations are probably not (yet) threatened, but quantitative data on their distribution and life-history traits in the mountain ecosystems are not available. In recent years (2008-2009), we started on a coordinate project between Germany and Italy, in order to compare larval performances and survival rates in different habitats and altitudes, and thereby trying to understand the main drivers of decline. In fact, in many butterfly species, the larval stages have more specific requirements than those of the adults, because they are less mobile. They are, therefore, strongly linked to the micro-environment occurring at the oviposition site and have longer 'life' than the adults (Albanese et al., 2008; Bergmann, 1999). Understanding larval requirements is essential for identifying what '*habitat quality*' means for the majority of butterflies, while understanding the larval needs of target species may be crucial to design adequate management practices (Ellis, 2003).

Colias palaeno is a spring-developing, but heliophilous, organism and hibernates as a small larva (2nd or 3rd instar). Therefore, this species may be particularly sensitive to the cooling of microclimates in spring, which can be determined by the seasonal advance of plant growth because of climatic warming and nitrogen deposition (Wallisdevries & van Swaay, 2006). Moreover, a specialized association of a monophagous larva with its host species may create an opportunity for climate change to impact on larval feeding (Dennis & Shreeve, 1991). Changes in soil moisture, soil temperature, precipitation, air temperature etc. can affect the nutrient content or palatability of the host plant. Any change in food quality may be harmful to specialised herbivores, since it may force them to increase foraging time, and thereby expose them to predators or increase the potential for food limitation (Ayres, 1993). Dury et al. (1998) have found changes in the digestibility of leaf materials, due to reduced leaf nutritional quality, as a result of reduction in foliar nitrogen concentration and parallel increase in condensed tannin content. In fact, plant secondary chemicals can constrain

herbivore growth by deterring consumption and by interfering with digestive efficiency (Ayres, 1993). Therefore, two separate factors may be potentially responsible for the decline of *C. palaeno* : i) changes in micro-environment structures, in particular in features influencing local micro-climate; ii) alterations in the general leaf quality of the larval host plant.

Some preliminary results obtained on Alpine populations suggest a dependence between polyphenols content and larval survival rate. Higher levels of secondary metabolites could have detrimental effects on larval performance, while lower amounts of phenolic compounds have been observed in the heath lining the timberline, a kind of environment not present in the lowlands area of great declined of South of Germany (*unpublished data*).

3.6 The functional hotspots of biodiversity - The myrmecophilous insect

Pre-adult myrmecophilous butterflies spend a variable amount of time living in association with ants. Most of them are commensal or mutualist organisms, either left undisturbed by ants, or becoming actively protected by them. In nature, we observe many different levels of myrmecophily, ranging from facultative (butterfly larvae are occasionally tended by ants while feeding on their food-plant, more rarely on aphids etc.), to obligate (butterfly larvae cannot survive unless they are taken into the ants' nests where they fulfil their trophic need and spend the climatically worst parts of the year; see Fiedler, 1991). Butterfly larvae, in fact, may be tended within their phytophagous (more rarely carnivorous) foraging areas, or inside the ants' colonies, depending on cases (Hölldobler & Wilson, 1990; Thomas et al., 2005). In Europe, *Plebejus argus* is among the relatively few obligate mutualistic myrmecophilous Lycaenids, and is associated to *Lasius niger* ants. Other myrmecophilous butterflies have evolved as ants' social parasites and most of them are rare in comparison to the abundance and distributions of their ant hosts (Hölldobler & Wilson, 1990; Thomas et al., 2005). Butterflies of the genus *Maculinea* are among the best documented examples of myrmecophilous insects. They are obligate parasites, strictly and only depending on the ants classified in the *Myrmica* genus. Larvae of these butterflies, after spending a short period while feeding on a specific foodplant, penetrate into the *Myrmica* ant-colonies, where they spend 11 to 22 months (Barbero et al., 2009a,b; Thomas et al., 1998a; Witek et al., 2006). Such a close and obligate interaction shelters myrmecophilous butterflies from seasonal climate variations, but leaves them vulnerable to large scale environmental changes, since their survival depends on the persistence of multiple factors, such as a strictly specific larval host plant (LHP) and a similarly species-specific ant. Most of them are rare and protected in Europe (e.g. Habitat Directive Annex II).

3.6.1 Host ant specificity

Regional populations of specialist insect species, such as myrmecophiles, may be adapted to their local environment more strongly, and at smaller scales (Schönrogge et al., 2006), than is recognized in current paradigms of host specificity. Across Europe investigations of the host-ant specificity of *Maculinea* butterflies have demonstrated that patterns are very complicated. Both single- and multiple-host populations are currently known to exist for each of *Maculinea* species, with the only exception of *M. nausithous*, which is associated to one *Myrmica* species (*M. rubra*), both at local and at broad geographical scale (Casacci et al., 2011; Patricelli et al., 2010; Pech et al., 2007; Tartally et al., 2008; Witek et al., 2008). Moreover, on the base of host specificity data obtained for *M. alcon* and *M. rebeli*, Pech et al.

(2007) postulated that single-host specialization tends to develop near the butterfly range edges. Thus, populations such as those of *M. rebeli* in Spain or in France, or of *M. arion* (Fig. 8) in England, may prove even more vulnerable to climate or habitat changes than other European populations.

Fig. 8. *Maculinea arion* on its alternative LHP – *Origanum vulgare.*

The survival of myrmecophilous insects depends on the interactions among several species, so that a precise understanding of host specificity patterns is required before we can set up any appropriate habitat management. This may be even more essential for single host butterfly populations, since they are adopted to an ant-host possessing narrow niche preferences, such as in the case of the *M. arion* populations living in England (Thomas et al., 2009).

3.6.2 Habitat fragmentation and dispersal ability

The myrmecophilous life style and its associated high host-specificity tend to select for a limited dispersal. In fact, *Maculinea* butterflies show very low mobility, compared to most other butterflies (Nowicki et al., 2005a) and distances covered by *Maculinea* individuals typically vary between 50 and 400 m, even though movements of over 5 km were sometimes observed in *M. arion* (Nowicki et al., 2005b). Many insect species are currently forced to live in fragmented populations, and occupy habitats which in the course of time became disrupted into small, isolated patches. Their low dispersal ability makes myrmecophilous butterflies more vulnerable to the effects of low genetic variability, generating high extinction probabilities for some particular subpopulations. Moreover, since they depend on host ants as well as on foodplant availability they are also more sensitive to habitat fragmentation, since their survival depends on the abundance and density of both resources on each particular habitat patch (Nowicki et al., 2007, 2009). Metapopulation studies carried out on *M. arion* in the Alps have shown that only sites with high density of Thymus (foodplant) and host-ant

(Myrmica sulcinodis) support high numbers of butterfly adults (Casacci et al., 2011). Similar effects of habitat quality were observed for *Plebejus argus*, whose persistence was determined by the presence and density of mutualistic ants (Seymour et al., 2003).

3.6.3 Land use and management regime

Apart from habitat fragmentation, another main cause for the decline of butterfly populations is in changes in land use. Most myrmecophilous butterflies are associated to some particular type of grassland, either hygrophilous or xerophilous depending on the species. The persistence in time of such habitats is necessarily man-dependent, since they will become invaded by trees and shrubs as soon as they will be abandoned. Many of them, as a consequence, were transferred to other types of land use, while some management regimes such as mowing or grazing were suspended. All across Europe, this resulted in strongly declining biodiversity, during the last decades (van Swaay, 2002). A classical example showing how changes in habitat management may lead to local extinctions relates to the native English populations of *M. arion*, which became extinct in 1979 (Thomas, 1980). In England, the survival of *M. arion* larvae depended on the presence and abundance of only one host ant - *M. sabuleti*. In N Europe, this ant species is associated with short-turf grasslands, and prefers warm and sunny places. Changes in habitat management related to the loss or strong decrease in grazing rabbits (disease), as well as the cessation of grazing by cattle and sheep, caused the short-turf grass to be replaced by longer turf, and resulted in a cooler soil microclimate. This, in turn, favoured other *Myrmica* species like *M. scabrinodis*, which displaced the main host ant *M. sabuleti*. After all these changes the survival of *M. arion* larvae decreased dramatically and finally led to population extinction (Thomas, 1980). Nowadays, thanks to a highly-organized management programme, *M. arion* was successfully reintroduced to its native area, starting from populations from S Sweden (Thomas et al., 2009).

Johst et al., (2006) presented a model showing how mowing regimes influence population persistence in two great blue butterfly species, i.e. *Maculinea teleius* and *M. nausithous*. Factors responsible for density-independent and density-dependent (myrmecophily) mortalities of larval stages were taken into account. Results indicated that mowing once per year, or even every second or third year, was the most suitable type of management for both *Maculinea* species.

3.6.4 Climate changes

It is well know that butterflies react to temperature rises by adapting their phenology to that of their larval host plant and by searching for suitably cooler places by shifting in latitude or altitude (Roy & Sparks, 2000; Stefanescu et al., 2003; Wilson et al., 2007; Settele et al., 2008). Changes in latitude have been observed in some species (e.g. *Polygonia c-album* see Thomas, 2005) but no changes have been demonstrated to occur in many others. Species such as *M. alcon*, *M. nausithous*, *M. teleius* are all strictly linked to the hygrophilous grasslands and, as mentioned above, they have low dispersal capability (Nowicki et al., 2009). Since they typically occupy lowland meadows, these *Maculinea* species will be probably scarcely prone to shift in altitude. It is interesting to notice that even though each *Maculinea* butterfly exploits a species-specific larval host plant at least in most of its range, some species can exploit a second host plant. We can speculate that, while a shift onto a congeneric food-plant might occur in connection with natural dispersion events, or the colonization of new habitats, a shift to a more taxonomically distant LHP might occur as a response to changes

in the phenology of the primary host plant. This could be the case for *M. arion* in Italy, where it ranges from the 200 m of some peninsular sites, to over 2000 m in the Alps, and occurs in a wide variety of habitats, from sub-Mediterranean to Alpine.

We can also speculate that, during the last glaciations, in the north of Italy Thymus-eating *M. arion* may have found refuge in the lowlands at the base of the glaciated Alps (Schmitt, 2009), while it may have survived at higher altitudes, in the hotter latitudes of central and southern Italy. After de-glaciation, following the natural spread of its host plants and ants, the Alps may have been re-colonized (Schmitt, 2009). At the same time, the rising temperatures on the already warm grasslands occupied during glaciations might have resulted in changes in LHP phenology. This may have led *M. arion* to either of two possible responses: i) in the places where, by chance, a taxonomically distant but phenologically similar plant, became successfully exploited, there was a change in LHP (*Origanum vulgare*), or ii) there where these changes were impossible, butterflies were forced to adapt their phenology to that of the Thymus plants (see Fig. 9). To partially support our supposition, we can observe that in each Italian site where *Maculinea arion* exploits *Origanum vulgare* (lowland populations) also Thymus is present, but has anticipated phenology. In these communities, the flight period of *M. arion* occurs at the same time as for the *Thymus*-eating populations occurring in the Alps. On the contrary, in those *M. arion* populations where no host plant shift was possible, butterflies adapted their phenology to their *Thymus* host plants, so that their flight period occurs much earlier than in the alpine, as well as the lowland population of the North of Italy.

Fig. 9. *Thymus* spp. *Maculinea arion* main LHP.

The current global climate warming may be expected to cause similar effects as those of past de-glaciation times. Were this the case, the conservation of the alpine populations of *M. arion,* where an altitudinal shift cannot occur, will be assured, in some areas, by the coexistence of the two possible larval host plants, or in other areas by the capacity of this butterfly to adapt its phenology to that of thyme. On the other hand, the future of those populations which became adapted to a second LHP (*O. vulgare*) will be assured by the butterflies' capacity to adapt their phenology, or otherwise by shifting onto a third host plant, as yet unknown. Before we can evaluate the butterfly's ability to adapt its phenology to that of the host plants, it will be fundamental to gain information on the possible effects of temperature on the phenology of the host-ants' nest. It is well know, in fact, that *Maculinea* larvae acquire most of their body mass after their winter diapause (Thomas et al., 1998a; Witek et al., 2011), i.e. at a time when the nest is active and food (ants' brood) is widely available. In the lack of a full understanding of temperature effects on nest activity, it will impossible to identify the break point between the butterfly's need to adapt to the LHP phenology, on the one hand, and to spend enough time inside the *Myrmica* nest to acquire sufficient body mass to conclude its larval cycle, on the other. Any failure in balancing between these needs will result in a major threat for all those population in which phenological shifts are already observable. An early emergence will be favoured only on those cases when larvae will have eaten enough to pupate early.

Butterflies of genus *Maculinea* have become 'flagships' of European biodiversity conservation (Thomas & Settele, 2004) and are perceived as umbrella species covering many grassland communities (Casacci et al., 2011; Randle et al., 2005; Spitzer et al., 2009). Special conservation efforts should be focused on those habitats where myrmecophilous insects are abundant, since protecting these sites will also allow us to protect many rare plant and invertebrate communities.

In a broader time frame, it will be important for us to gain precise knowledge on the ecological requirements of myrmecophiles under current climate and habitat change scenarios, in order to try and reverse current declines, by restoring optimum ecological conditions (Thomas et al., 2009). Management activities suitable for creating cooler micro-topographies and cooler successional stages will mitigate the impact of climate warming both in the short- and in the intermediate- term.

4. Conclusions

Species' extinctions are rapidly increasing everywhere in the world, as well as in Europe (Hobbs & Mooney, 1998). Species loss proceeds by increasing the extinction rates of individual populations, rather than by sudden and complete species losses (Ehrlich, 1994). Species, in fact, may lose a lot of their local components (i.e. populations) before they become threatened or even scarce in nature and population decline is a reversible event, while species' extinctions are not. So that detecting, understanding and halting this trend must be central in national and regional conservation planning.

Butterflies have attracted wide public attention owing to their endangered status, their beauty as adults and their extraordinary life histories, thereby becoming a flagship group. Their short life-cycle is sometimes very complex e.g. in the case of myrmecophiles. Butterflies have well-known ecological preferences and quickly respond to the action of drivers of change even more and faster than other well-known taxa like birds and vascular plants (Thomas et al., 2004; Warren et al., 2001). In addition they are a good indicator group

for other insects taxa (Thomas, 2005; van Swaay et al., 2010), but at least in Europe they are increasingly restricted to a small and fragmented portion of landscape.

For centuries most of Europe's land has been used by humans to produce food, timber and fuel and provide living space, so that currently in western Europe more than 80% of the land is under some form of direct management (European Environment Agency, 2010). Consequently, European species are to a large extent dependent upon semi-natural habitats created and maintained by human activity, particularly on traditional, non-intensive forms of land management. These habitats are under pressure from agricultural intensification, urban sprawl, infrastructure development, land abandonment, or desertification. Many species are directly affected by overexploitation, persecution and impacts of alien invasive species, as well as by climate change being set to become an increasingly serious threat in the future. Although considerable efforts have been made to protect and conserve European habitats and species diversity (e.g. Butterfly Prime Areas, Red Data Book, NATURA-2000 Network etc.) decline continues to be a major concern in the region.

Because of its geographical position and latitudinal and altitudinal gradients, Italy plays a central role in biodiversity conservation in this changing scenario.

5. Acknowledgements

6. References

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Changing Diversity in Changing Environment
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As everybody knows, the dynamic interactions between biotic and abiotic factors, as well as the anthropic ones, considerably affect global climate changes and consequently biology, ecology and distribution of life forms of our planet. These important natural events affect all ecosystems, causing important changes on biodiversity. Systematic and phylogenetic studies, biogeographic distribution analysis and evaluations of diversity richness are focal topics of this book written by international experts, some even considering economical effects and future perspectives on the managing and conservation plans.

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