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Special paper

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MITES (ACARI) AS INDICATORS OF SOIL BIODIVERSITY AND LAND USE MONITORING: A REVIEW

ABSTRACT: Soil is an important component for monitoring of sustainability of land use in relation to both the conservation of natural resources and biodiversity of ecosystems. Recently research has focused on the role of mites (Acari) in biomonitoring and their importance as soil bioindicators. Mite communities are extremely sensitive to all types of soil disturbance. This article presents a review of some studies on mite communities in order to discuss whether the diversity and numbers of mites present in the soil can reflect human impact on ecosystems and landscape, and whether mites can be used in monitoring systems. Limited data available about mite communities in agricultural ecosystems (in use and abandoned) in SW Norway from author's own studies (published and in prep.) are compared with data from *ca.* 55 papers (mostly concerning Europe).

IT was concluded that: (1) Most oribatid mites with their long life span, low fecundity, slow development and low dispersion ability can be a robust indicator of the environment. Changes in the dominance structure of mite communities (Oribatida to Actinedida ratio) are suggested to be an 'early warning' criterion for stressed mite communities. (2) Both the number of species and the percentage abundance of Nothroidea and Ptyctimina decrease following human impact on the landscape. Even if individuals cannot be determined to species levels (e.g. Ptyctimina), the percent contribution and

frequency of these taxa in relation to stress gradient (input levels) within the landscape provide valuable data. (3) *Tectocepheus velatus* (Michael 1880) and other oribatid taxa with a similar life-history strategy should be evaluated as potential bioindicators for impoverished ecosystems. (4) National and local reference data sets on the biodiversity of mite communities in diverse habitats and along stress gradients need to be collected. (5) Residual natural and semi-natural habitats (such as old woodlands, riparian ecosystems, old hedges and grasslands) with species-rich mite communities found in rural and urban landscapes should be preserved as refuges for dispersion of soil fauna. (6) Comparison of mite communities in traditional, low-input farmland in Norway with those from human-dominated landscape in other European countries can contribute to a better understanding of how human activity alters biodiversity along land-use gradients. This will aid the development of a soil bioindicator system. (7) There is a need to develop standardised procedures for the collection of samples and analyses of data sets adapted to ecological soil acarology.

KEY WORDS: biodiversity, soil fauna, monitoring, disturbance, Acari, Oribatida, Actinedida, Gamasida, Uropodina

1. INTRODUCTION

In recent years, several initiatives have been started in European countries and worldwide designed to apply the Convention on Biological Diversity into practice in order to control a significant reduction of the current rate of biodiversity loss at the global, regional and national levels. Consequently, to develop biodiversity indicators as the tools for the assessment of sustainability of terrestrial ecosystems, including the soil, became necessary in the last decade and a number of scientific papers, mainly reviews, discussing the bioindicator systems were produced (Büchs 2003, Balmford *et al.* 2005, Beck *et al.* 2005, Breure *et al.* 2005, Buckland *et al.* 2005, Dudley *et al.* 2005, Römbke *et al.* 2005).

In all European countries, systems for soil monitoring with a focus on the physical properties (structure, texture, moisture) and chemical dimensions (soil nutrients, organic matter, pH etc.) have been already established (Øygarden and Grønlund 2003, Breure *et al.* 2005). For instance in the Netherlands, a total of 200 locations representing ten characteristic combinations of soil type and land use in the rural part of the country have been selected for monitoring (de Goede and Brussaard 2002). Initially, soil analyses were focused primarily on chemical properties of the soil, but soil organisms, including mites, have been monitored in this country since 1993. The data were used to calculate a *soil quality index* facilitating ecological soil assessment (de Goede and Brussaard 2002). Recently, preserving soil biota is perhaps the most important and challenging task for sustainability of land use (Hågvar 1998a, Bengtsson *et al.* 2000, de Goede and Brussaard 2002).

Monitoring of biodiversity can be met in various ways. It may be studied from a habitat perspective (e.g. agrosystems), the input of human stress factors (e.g. chemical pollution), or biodiversity of given taxa (e.g. birds). This article focuses on the soil, as it is the important component for monitoring of sustainable land use in relation to both the conservation of natural resources and monitoring of biodiversity. In this paper, selected aspects of ecological indexes for soil quality

based on the mite (Acari) communities are discussed.

The use of mites for monitoring of the soil systems is a new area of research within the field of acarology. Acarologists are confronted with many difficulties. Some of these problems are discussed in the paper. In addition, diversity of taxa given in *ca.* 55 published papers is reviewed to show the usefulness of mites as bioindicators. Only papers that present the abundances such that could be counted by author as ratios and/or percentages are used. Species richness is reviewed only for taxa that are comparable, i.e. all collected specimens have been determined to species level. Available data on the mite communities in agricultural ecosystems (in use and abandoned) in south-west Norway from author's studies (both published and unpublished) are compared with data from other European countries.

2. BIODIVERSITY OF MITES AND THEIR ROLE IN THE SOIL

2.1. Taxonomic diversity of mites

Mites are small arthropods (microarthropods which belong to soil mesofauna). They are commonly 0.1–1 mm in length and found throughout the planet in every kind of habitat from the sea to freshwater, and in all terrestrial habitats e.g. in soil, tree canopy, rotting woods, tree hollows, nests of vertebrates, dung and anthills. Mites are classified into following suborders: Gamasida, Actinedida, Acaridida and, in addition to the well known Ixodida [with sheep tick, *Ixodes ricinus* (Linnaeus 1758)], the most abundant suborder Oribatida. Krantz (1978) gives the details on collection, extraction, preservation and determination of mites. In most studies, the scientists use the Berlese-Tullgren funnel to extract microarthropods from soil samples (Ducarme *et al.* 2002) because of its simplicity and short processing time. The common size of soil samples is often about 5 cm in diameter and 5 cm depth. Most mites in native soils occur in the top soil horizons.

Nomenclature and systematics of mites were changed during years, and different names for the same taxon (Krantz 1978) are

used, therefore common synonyms are given below.

Gamasida (or Mesostigmata) are usually represented by Gamasina and Uropodina. Gamasida are embedded at all levels of soil food webs, and they play many ecological roles within the ecosystems. Some are parasites [for example *Ornithonyssus bacoti* (Hirst 1913)], others, e.g. most Uropodina, feed predominantly on decomposing litter and fungi. But most Gamasina are mobile predators which prey on Enchytraeidae, Nematoda, Collembola, insects and mites (particularly on weakly sclerotized Actinedida, Acaridida and Oribatida juveniles) (Koehler 1999). Mites from the Phytoseiidae family are the most common predators of phytophagous mites found on most plant species. Some species are widely studied and used as the biological controls for mite pests (Edland 1994). Rhodacaridae, Macrochelidae and other Gamasina also play an important role in the control of *Rhizoglyphus* (Acaridida), Onychiuridae (Collembola), nematodes and flies (Koehler 1999). In sustainable and stable ecosystems, euedaphic (e.g. Rhodacaridae), hemiedaphic and epigaeic species (Macrochelidae, Parasitidae and Veigaiidae) must be present (Koehler 1999). Many Uropodina and Gamasina rapidly colonize new habitats by phoresy (Evans *et al.* 1961, Siepel 1994, Koehler 1999). There are 220 reported species in Norway (Gwiazdowicz and Gulvik 2005a), whereas there are over 1000 known species in Central Europe (Koehler 1999).

Actinedida (or Prostigmata) is a large group with soft-bodied to heavily sclerotized species. These mites are red, brown, green, orange or whitish, and are the most heterogeneous group with respect to life-history strategies and feeding habits (Krantz 1978, Siepel 1994). The larval stages of Parasitengonae, including the well known water mites (Halacaroidea), are ectoparasites of arthropods. Opiliones are often infested with *Leptus* larvae. In spring and autumn, one can often see large, red post-larval stages of Parasitengonae on open ground in gardens where they prey on small arthropods. Some Actinedida are phytophagous pest species (Tetranychidae, Eryophyidae) and together with the honeybee tracheal mite *Acarapis woodi*

(Rennie 1921) are economic pest. Most Actinedida, e.g. Tydeidae, are polyphages. About 384 species have been recorded in Norway (Mehl 1979).

Acaridida (or Astigmata) are soft-bodied, primarily saprophagous, fungivorous or graminivorous mites. They are common in arable fields fertilised by manure, and they also often occur in stored and processed foods (examples include the genus *Acarus*, *Tyrophagus* and *Rhysoglyphus*). Wasylik (1986) reported 41 200 individuals m⁻² of acaridids in an arable field.

Oribatida (or Cryptostigmata), called moss mites or beetle mites, are free-living, slow-moving, heavy sclerotized mites which are found in all terrestrial habitats. They are the most numerous (accounting for 60–90% of all mites at a given site), and probably the most species-rich mite group. There are about 7 000 species worldwide, over 1 000 species in Europe and over 500 species in Poland recorded (Olszanski *et al.* 1996). However, they have rarely been a subject of study in Norway. Oribatida are classified into two major groups, Macropylina (to this group Nothroidea and Ptyctimina belong) and Brachypylina. Brachypylina is further subdivided into Gymnonota and Poronota (or Apterogasterina and Pterogasterina according to other nomenclature) depending on the presence of pteromorphs (Krantz 1978, Balogh and Balogh 2002). Mehl (1979) reported in Norway 244 oribatid species included 27 Nothroidea (genus *Camisia*, *Nothrus*, *Platynothrus*, *Heminothrus*, *Trimalaconothrus* and *Malaconothrus*) and 23 Ptyctimina species (ptychoid oribatid mites, the so-called box mites).

Most Oribatida, e.g. all Nothroidea, are *K*-selected persisters, but some taxa are *r*-selected colonizers, e.g. *Tectocephus velatus* (Michael 1880) and *Oppiella nova* (Oudemans 1902) (Norton 1990, Siepel 1994). Dispersal experiments indicate that most Oribatida species have the potential for land colonisation over a distance of 30 m in a time span of 30 years (Ojala and Huhta 2001). Some species exhibit a wide zoogeographical distribution, e.g. *T. velatus* and *Platynothrus peltifer* (C. L. Koch 1839). Oribatids are of veterinary importance as intermediate hosts and vectors of tapeworms (Helle 1971).

2.2. Mites in soil food web

The high α -diversity of decomposer invertebrates in soil is one of the great challenge of soil biology (Anderson 1975, Giller 1996, Maraun *et al.* 2003.). As soil communities are extremely rich in species diversity, they have often been described as the 'poor man's tropical rainforests' (Usher *et al.* 1979, Giller 1996). Explanations for high diversity have traditionally been based on the equilibrium concept of community structure and a high degree of resource/niche partitioning (Giller 1996). Traditional niche theory would predict that greater biodiversity should result in more specialized niches (smaller niche width) and a greater degree of resource partitioning (i.e. reduced niche overlap) (Giller 1996). Previously, it has been assumed that oribatid mites are generalist feeders with wide feeding habits and weak trophic niche differentiation. But recently, Schneider *et al.* (2004) documented strong trophic niche differentiation within Oribatida in field-collected animals, and this spans three to four trophic levels. Differential feeding of morphologically similar species, such as *Nothrus silvestris* (Nicolett 1855) and *N. palustris* (C.L. Koch 1839) contribute to the co-existence of many species and may confirm the niche theory. Most species are probably secondary decomposers (in contrast to fungi and bacteria that are considered as primary decomposers). This aspect of nutrition has been termed the 'peanut butter sandwich' problem (cited after Johnston and Crossley 1993) owing to the fact that fungal hyphae are in-grown into the decaying substrate and many Oribatida feed both on fungi and decaying plant material. Xylophagous Ptyctimina feed on the woody tissue of plants (primary decomposers) and have been found in conifer needles, leaves, twigs and rotting branches (Krantz 1978, Hågvar 1998b). Fungivorous mites (secondary decomposers) may feed on fungal cell contents only (piercing mites – browsers – mainly Actinedida) or both fungal cell contents and cell walls (chewing mites – grazers – mainly Oribatida and fungivorous Gamasida) (Siepel and de Ruiter-Dijkman 1993, Didden *et al.* 1994, Hu-

bert *et al.* 2001). In experiments, the mites are able to survive on various types of food; it is possible that they shift feeding habits as resources change. In 1990's, the oribatids have been divided into feeding guilds on the basis of their enzyme activities (Siepel and de Ruiter-Dijkman 1993, Hubert *et al.* 2001). More details on the feeding ecology are presented by Behan-Pelletier (1999) and Huhta (2007).

Mites make up a large and functionally important part of the mesofauna in the soil; this makes them suitable bioindicators for soil system. A nation-wide study of soil biota in Great Britain suggested that Acari are the most frequently recorded group, occurring in 94% of all soil samples (Black *et al.* 2003). Mites influence the decomposition rates and plant growth (Schulz and Scheu 1994, Siepel and Maaskamp 1994, Hubert *et al.* 2001, Bradford *et al.* 2007, Huhta 2007), disperse microorganisms and fungi (Renker *et al.* 2005) and affect growth and species composition of fungi by selective grazing (Mc Gonigle 1995, Maraun *et al.* 1998, Schneider and Maraun 2005, Schneider *et al.* 2005). However, the relationship of particular species to soil function such as decomposition and nutrient cycling, is often unclear (Bengtsson 1998, Behan-Pelletier and Newton 1999).

With respect to the food web components, soil fungal-microarthropod systems contribute considerably to nutrient cycling in agricultural soils with low mineral N fertilisation as well as in forests and semi-natural grasslands (Hendrix *et al.* 1986, Beare *et al.* 1992, Pokarzhevskii *et al.* 2000, Hedlund *et al.* 2004). Thus, the fungi-fungivore interactions are important in regulating litter N dynamics in surface litter (Beare *et al.* 1992). The contribution of fungi to formation of soil aggregates may represent an important biotically regulated mechanism for the protection of soil organic matter (Beare *et al.* 1997a, Guggenberger *et al.* 1999). It is provided that good soil protection and high levels of soil organic matter favour higher biodiversity. Therefore, the standing stock of organic matter in the soil is an important determinant of microarthropod abundances at a local scale (Blair *et al.* 1994, Mäder *et al.* 2002). The development of fungal-micro-

arthropod systems in impoverished soils appears to progress very slowly (Werner and Dindal 1990, Scheu and Schulz 1996). In contrast, the bacterial-algal-protzoan channel dominates in the soil of agrosystems under intensive tillage (Hendrix *et al.* 1986, Brussaard *et al.* 1990, Guggenberger *et al.* 1999, Pokarzhevskii *et al.* 2000). However, an increase of soil microbial biomass or activity cannot automatically be interpreted as an increase of soil quality. Since different abiotic factors, such as meteorological conditions (temperature and precipitation), change rapidly fungal and microbial biomass and soil respiration, soil microorganisms respond very rapidly to these environment changes (Schnürer *et al.* 1986, Gomoryova 2004). It can be seen that, in order to correctly assess the state of soil across sites and habitats, it is necessary to incorporate biological monitoring of soil fauna. Many acarologists have proposed the mites as bio-indicators of environmental quality (Moore and DeRuiter 1993, Paoletti and Bressan 1996, van Straalen 1998, Behan-Pelletier 1999, Paoletti 1999, Maraun and Scheu 2000, Behan-Pelletier 2003, Ruf and Beck 2005).

2.3. Effect of human activities on mite communities

Soil biotic systems in highly and frequently disturbed ecosystems (such as intensive agricultures) are found to be in an early stage of permanent secondary succession (Bengtsson *et al.* 2000). Many studies have shown that all types of human disturbances decrease the abundance and species richness of mites, especially oribatids. For example, Niedbała *et al.* (1990) reported that soil in deciduous forest was inhabited by 37 species of Actinedida, 56 of Oribatida and 25 of Gamasida species. The relevant numbers from the parks in Warsaw were 31 of Actinedida, 14 – Oribatida and 15 species – Gamasida, whereas 27 species of Actinedida, 9 – Oribatida and 15 – Gamasida were found in the green areas within housing estates in Warsaw (Niedbała *et al.* 1990). Seventy species of Oribatida were found in four residual woodlands in Italy, whereas

nine maize monocultures were inhabited by only half of these number (Paoletti 1988). Crossley *et al.* (1992) reported that 24 genera of oribatids occurred initially in soil before cultivation. Two years after establishment of arable fields, 15 of these genera persisted in no-tillage plots and only five genera in conventional (high input) tillage fields. Densities of mites increased in agrosystems converted from conventional to an integrated system (Didden *et al.* 1994). It is well established that most of mites (Acaridida, Uropodina, fungivorous Actinedida, some Gamasina and most of the Oribatida) prefer habitats rich in organic matter. The most important factors responsible for the development of oribatid mite communities during secondary succession on fallow was the formation of a mineral soil rich in humus accompanied by a low soil density and high water content (Scheu and Schulz 1996).

The loss of biodiversity described in agricultural lands also concerns the effects of modern forestry. Intensive nutrient fertilisation of forests has been proposed as a method to increase production of biofuels as a replacement for fossil fuels. Recent studies in Sweden indicate that intensive forest fertilisation could cause large shifts in soil microarthropods (especially Oribatida and Collembola) communities, but the risk of species loss will probably depend upon the size of the areas used for this purpose (Lindberg and Persson 2004). The addition of lime to the soil reduced the abundance of various Oribatida and some Gamasida species in coniferous forest (Hågvar and Amundsen 1981). Different harvesting practices also influenced on the soil fauna. Removal of harvest residues had negative effects on the abundances of animals at higher trophic positions and abundance of predators such as gamasids which decreased (Bengtsson *et al.* 1998). Battigelli *et al.* (2004) found that a short-term impact of forestry on forest soil caused decrease of the density of soil mesofauna, in general, and oribatid mite species, in particular. Stem-only harvesting reduced total mesofauna densities by 20% relatively to uncut forest values, whereas combination of whole-tree harvest, forest floor removal and heavy soil compaction significantly reduced total soil mesofauna densities by 93% com-

pared to the uncut forest control (Battigelli *et al.* 2004). Bengtsson *et al.* (2000) argue that the next generation of forestry practices needs to understand and follows natural disturbance dynamics much better than the present ones.

Siepel (1995, 1996) provides evidence that the decline of biodiversity is not a random loss of species but follows a predictable pattern. A key to the life-history tactics (numbered from I to XII) based on multi-dimensional form derived from dispersal and reproduction traits, synchronization of life cycles etc. have been developed by this author (Siepel 1994, 1995). The life-history tactics system was applied for microarthropods. Siepel (1996) showed that the main differences between forest sites and low-input grasslands were manifested as the decreasing fraction of short-living thelytokously reproducing microarthropods (tactic X) and absence of long-living thelytokously reproducing microarthropods (tactic IX) in the grasslands. The differences between low-input and high-input grasslands were best characterized by the enormous increase in the fraction of phoretic species (tactics II or IV) in high-input grasslands, and a revival of very few species with thelytokously reproduction. The highest fraction of species with sexual reproduction [such as *Liebstadia similis* (Michael 1888), tactic XI] has been found in low-input grasslands whereas the highest fraction of long-living thelytokous species (tactic IX, for example nothroid species *Platynothrus peltifer*) – in the forests (Siepel 1996). The use of microarthropod life-history tactics system permits the comparison of the effects of management in different biotopes and countries (Siepel 1995).

3. BIOINDICATORY IMPORTANCE OF COMPOSITION AND DIVERSITY OF MITE COMMUNITIES

According to Crossley *et al.* (1992), Bengtsson (1994), Siepel (1995), van Straalen (1998), Kimberling *et al.* (2001) and Buckland *et al.* (2005), the following indices can be used to measure the diversity and composition of mite communities:

1) Species diversity and species similarity [for example Shannon-Weaver or Shannon-

Wiener, and Sørensen's indexes, log-normal distribution of species, Renkonen index];

2) Distribution of higher taxa (e.g. densities per square meter of soil, percentage contribution of the taxa or ratios between two taxa, for example ratio of Oribatida abundance to Actinedida abundance);

3) Distribution of functional groups (for example different feeding guilds, life-history strategies, dispersal strategies, morphological or ecological morphotypes).

To use mites for ecological studies, a large number of methodological difficulties need to be overcome. Often it is difficult to make the comparisons between data sets from different papers, if the lists and abundance of taxa for each site are not given. In this paper, these and other problems will be exemplified using Gamasina and Oribatida data.

3.1. Sampling problems – minimal vs. optimal data sets

In order to identify the mites indicator value, one must have a 'negative' (e.g. disturbed sites) and 'positive' reference datasets (undisturbed sites) (Beck *et al.* 2005). For positive reference areas within a given site type, native areas (close-to-pristine ecosystems) should be chosen. In practice, nearby situated forests or/and semi-natural grasslands are often used in comparison with arable land or urban habitats to establish a range of values that are site-specific. Knowledge of the 'normal' state of the reference habitat needs to be established by measuring ecological indicators over time at a smaller scale (e.g. for farmland and regional conservation purposes) and at a country-wide level (e.g. national conservation of spatial biodiversity). Reference collections should be normally restricted to sites having similar biological, chemical, and physical conditions, and should take into account the geographic and climatic conditions. Until today, there is insufficient knowledge about the 'normal state' for native habitat types in Europe.

Legal initiatives have been undertaken to direct the research towards the implementation of reproducible and standardized methods to determine the ecological quality of soil (Römbke *et al.* 2005, Römbke *et al.* 2006). Ruf and Beck (2005) recommend

for monitoring purpose, to take 10 replicate samples in order to estimate the population density relatively reliable. To avoid seasonal or weather-dependent extreme deviations of community parameters such as the presence and absence of species, it is recommended that a minimum two sample sets should be collected per year. The optimum is the collection of four sample sets: at spring and autumn of two successive years.

In the context of point counts (single location and time-points), comparison of counts across species are invalid, because different species have different detectabilities (Niedbała 2000). The knowledge of species number is likely to be based commonly on a series of randomly placed samples, thus conclusions are drawn from a continuous and uniform habitat which in reality does not exist. A routine examination of about 20–30 cores taken from a single site should probably give a lower number of species in each of the ecosystems mentioned (α -diversity). The number of species increases with the number of samples collected, levelling out after about 25–30 samples. The greater is the variability within the site, the larger is the number of measurements that are necessary to obtain a representative value of species richness at the site. But rare patchy species will often be missed altogether by sampling, or even if found, will not always be recognizable as patchy unless the number of samples is very large.

3.2. Taxonomic dilemma of mites

The first step when using indexes based on species richness is to determine all adults to species level (most immature stages are not described). However, gamasids and oribatids are not identifiable without slide mounting because most of the specimens require examination of fine taxonomic details under the microscope and the use of relevant keys. Additionally, the identification of all the adults is not possible for some gamasids and oribatids because of their incomplete taxonomy. For some taxa, there are very few (as few as two) specialists worldwide who can determine individuals to species level. The consequence of these difficulties is that case studies vary in qualitative data regard-

ing species richness from one publication to the other. For example, Huhta and Niemi (2003) have not determined *Suctobelbella* to species level, Hubert *et al.* (2004) have not determined Oppiidae and four genera, while Paoletti (1988) has not determined seven genera to species level. Thus the indexes based on the presence-absence of species in case studies presented in the literature are not always comparable. In the most extreme cases, specimens are only determined to 'morpho-species' in order to overcome the well-known difficulties in taxonomic analysis at species level (Horwood and Butt 2000, Proctor *et al.* 2002, Kounda-Kiki *et al.* 2004). The morphospecies assignment within many mesostigmatid and oribatid mites corresponds well with known families or genera. Thus, species richness based on morphotypes does not reflect the true species biodiversity.

Noticeably, there is a conflict between systematists and ecologists (Behan-Pelletier and Newton 1999, Niedbała 2000). The systematists devote little attention to the ecology of individuals and populations, or to the organization of species assemblages in ecosystems. In contrast, soil ecologists are most interested in ecosystem processes, such as energy flux, decomposition, and nutrient cycling. They devote little attention to species richness and rarely study the ecology of individual species living in the soil. It is suggested that changes in diversity at coarse levels of higher taxa like families or orders are more likely to influence ecosystem function than changes at finer levels of resolution like species or genera (Beare *et al.* 1997b). It is relatively few studies where mites have been identified to species level (Ducarme *et al.* 2002). Bengtsson (1998) proposed that studies should focus on keystone species and function groups, rather than on species diversity.

Other problems are connected with a fact that the indexes based on species richness need to be constantly revised because a new viewpoint on the classification of a given species may arise and to account for newly discovered species. For example, Weigman (2002) concluded, after large-scale studies on many populations of genus *Tectocepheus*, that *T. sarekensis* (Trägårdh 1910) are subspecies of *T. velatus*. Another example where the sta-

tus of a species has been revised is *Camisia biurus* (C. L. Koch 1839) which is considered a synonym for *C. segnis* (Hermann 1804) (Olszanowski *et al.* 1996). Małkol (2005) proposed synonyms of seven Trombidiidae species and excluded two species from the checklist of trombidiids occurring in Poland due to misidentifications. What species have been recorded in given case studies is common question. For example, Rajska (1961) reported the occurrence of *Nothrus silvestris* var. *anauniensis* (Canestrini et Franzago) and *N. biciliatus* (C. L. Koch), but these specimens could have been *N. anauniensis* (Canestrini et Franzago 1876), *N. silvestris* (Nicolet 1855) and/or *Camisia biurus* (C. L. Koch 1839). Thus, the use of species richness recorded in papers to make a comparison between data sets is very limited.

Until now, mite fauna in Norway is poorly investigated and basic data about richness and distribution of species across regions and ecosystems are missing. Knowledge of the taxonomy and distribution of Gamasina in Central Europe (especially Germany and Poland) is much better documented than in other European regions (Ruf and Beck 2005). The taxonomic recognition of the oribatid mites in Europe is worse (Ruf and Beck 2005), however fauna of Oribatida in Poland is relatively well known (Olszanowski *et al.* 1996). Effort to gain knowledge about species distributions related to climatic and geographical factors, land-use systems, soil and vegetation types are laborious and require a close collaborative effort between systematists of different taxa and ecologists.

In practice, it seems that it is not possible to monitor the ecosystems using the indexes based on qualitative datasets about species richness of mites; moreover it is probably not possible to map and monitor the species richness of mites at national level.

3.3. Problem of rare species in mite communities

Natural, stable communities are generally characterized by having many rare species and few abundant ones (Hågvar 1994, Koehler 1999). If individuals per species

are presented on a logarithmic scale, the species numbers usually show a normal distribution, giving the so-called log-normal distribution. In ecosystems under stress, certain species become very abundant, and the log-normal curve is distorted (Hågvar 1994, Madej 2004). In practice, unstable ecosystems with low species number and low abundances (such as arable fields, heavy grazed grassland and early stages of primary succession) can often be dominated by a few species belonging to Gamasina, Actinedida and Oribatida (e.g. Paoletti 1988). These species have short life spans and high fecundity. Some of these species complete their life cycle in 4–7 days, leading to rapid increase of their abundance after any disturbance (so-called *r*-selected species). Such species can respond rapidly to nutrient pulses in their habitat (Siepel 1995, Koehler 1997, Behan-Pelletier 1999). In practice, 55% of all Oribatida in one maize monoculture was composed of only one species, whereas in four residual woodland (*Quercus-Carpinetum*), the most abundant species constitute only 20–30% of abundance of all Oribatida (Paoletti 1988). In urban green belts, 31% of all Gamasida individuals were composed of only one species, whereas in *Tilio-Carpinetum*, the three most dominant species together constituted only 30% of all Gamasida present (Niedbała *et al.* 1990). A similar trend was observed for several highly disturbed habitats and semi-natural deciduous forests in other countries (Sgardelis and Usher 1994, Scheu and Schulz 1996, Maraun and Scheu 2000, Clapperton *et al.* 2002).

Comparative analysis of mite communities from large data sets or field research papers (Bengtsson 1994, Maraun and Scheu 2000, Bengtsson *et al.* 2005) are probably only possible when regarding higher taxa, functional groups or some common and well-separated species. Accidental species should be excluded from these comparisons as it can lead to the possibility of unpredictable and undesirable properties of any index (Buckland *et al.* 2005). Usually, most of the species occur in less than 50% of samples taken from particular site and they can be found usually as very few individuals. It makes a problem when the

data are analysed statistically. There are no standard procedures for the treatment of rare species in mite data sets. Bengtsson (1994) excluded species which were present on average below 1 individual per soil sample from analysis of 67 forest sites. Huhta and Niemi (2003) discarded from statistical analysis all species present in numbers less than 30 for Gamasida or 40 for Oribatida in the whole material. Battigelli *et al.* (2004) considered only taxa which represent $\geq 1\%$ of all individuals in the whole material collected during four years from six plots. In 120 soil cores taken from four patches in a recently restored wooded meadow (old meadow with scattered pollarded trees, in fallow since 1970–80, restored in 1992), where 1267 Gamasina adults have been found, 35 species have been recorded (45 individuals were classified only to the genus level) (Gulvik and Madej unpublished). In each patch, only 2–5 species provided 5% or more of the total numbers of Gamasina. Similarly, among 50 Oribatida species recorded in this meadow, only five species made each $\geq 5\%$ of all Oribatida (Seniczak *et al.* 2007). Buryn and Hartmann (1992) recorded 72 species of Gamasina in a hedge (three patches) and in a surrounding intensively managed meadow (two patches); however only 14 of these species accounted for at least 5% of the total numbers of Gamasina per each patch.

A change in the dominance structure of mite community should be considered as an ‘early warning’ criterion for stressed mite communities. Descriptive statistics can be done on coarse taxa (for example total Gamasina) or only on the dominant, well-defined species.

3.4. Life-history traits of Gamasina

Ruf (1998) has proposed a system of soil quality classification based on biological criteria, called the *Maturity Index* (MI). This system is based on ranking gamasid mite taxa according to their life-history traits on an *r/K* scale (with values ranging from 1 to 4 for *r*-strategy and from 1 to 3 for *K*-strategy). The less disturbed soil, the value of the index is greater. This is caused by higher proportion of species with

predominantly *K*-attributes than those with high value of *r*-attributes. The MI is a sensitive bioindication tool and seems to be a good indicator of environmental quality in forest soils (Ruf *et al.* 2003). According to this range system, Rhodacaridae is seen as a ‘moderately’ *K*-selected family (ranged as 2). However, this classification is opposite to the fact that in disturbed habitats, *Rhodacarellus silesiacus* (Willmann 1936) occurs in large numbers. For example, this species made up 17% of all Gamasina found in six grasslands (Davis 1963) and was numerous on post-industrial dumps under primary succession (Madej 2004, Koehler 2000). According to Koehler (1997), this species is common in arable fields, early succession of rural sites, and in sandy soils low in organic matter. It seems that the data basis for the *Maturity Index* is not as well developed as it could be suggested by author, or there is methodology problem connected with the indicator value of the families. Future research is needed to test whether the MI for mesostigmatid mites is consistent for different ecosystems. Ruf (1998) and Čoja and Bruckner (2006) presented the use of MI only for forest habitats. The ranking of gamasid mite taxa (e.g. Rhodacaridae) should especially be re-evaluated.

3.5. Spatial and temporal variation of species diversity

The species diversity between individual studies varies widely in spatial and temporal scales. This claim is true first and foremost for *r*-selected species, such as many Gamasina and Actinedida. But *K*-selected species have also unpredictable distribution. To illustrate how occurrence of the species considered as typical for given habitat types varies across ecosystems, some data about Gamasina species from Europe are showed in Table 1. In addition, data from research made in two traditional farms in SW Norway (Slomian *et al.* 2005, Gwiazdowicz and Gulvik 2005a, Gulvik and Madej unpublished) were also used.

The genus *Alliphis* (in Europe represented commonly by *A. halleri* or *A. siculus*, Table 1) is considered as an extreme

Table 1. Occurrence of selected Gamasina species. Data from literature and author's own investigations. Abbreviations: P. alp: *Paragamasus alpestris* (Berlese, 1904); P. cam: *P. cambriensis* Bhattacharyya, 1963; P. cel: *P. celticus* Bhattacharyya, 1963; P. div: *P. diversus* (Halbert, 1915); P. hom: *P. homopodoides* Athias-Henriot, 1967; P. ins: *P. insertus* (Micherdzinski, 1969); P. lap: *P. lapponicus* (Trägårdh, 1910); P. mis: *P. misellus* (Berlese, 1903); P. run: *P. runcatellus* (Berlese, 1903); P. vag: *P. vagabundus* (Karg, 1968); A. hal: *Alliphis halleri* (G. et R. Canestrini, 1881); A. sic: *A. siculus* (Oudemans, 1905).

Species	Reference										
	Davis (1963)	Gulvik and Madje (unpubl.)	Madaj (2004)	Niedbala <i>et al.</i> (1981)		Paolletti (1988)		Seniczak <i>et al.</i> (1991b)	Stomian (2002)	Stomian <i>et al.</i> (2005)	
	Habitat type										
	Meadow (1)*	Restored meadow (1)*	Post-industrial dumps (3)*	<i>Tilio-Carpinetum</i> (1)*	Meadow (1)*	Arable field (1)*	<i>Quercus-Carpinetum</i> (4)*	Maize cultures (9)*	Tree belts (3)*	Deciduous forest reserve (1)*	Two traditional farms (3)*
P. alp	-	x	x	-	-	-	-	-	-	-	x
P. cam	-	x	x	-	-	-	-	-	-	-	x
P. cel	-	-	-	-	-	-	-	-	-	-	x
P. div	-	-	x	-	-	-	-	-	-	-	-
P. hom	-	-	x	-	-	-	-	-	-	-	-
P. ins	-	-	-	-	-	-	-	-	-	-	x
P. lap	-	x	x	-	-	-	-	-	-	-	x
P. mis	-	-	x	x	x	x	-	-	x	-	-
P. run	-	x	x	-	-	-	-	-	x	x	x
P. vag	-	-	x	-	-	-	-	-	-	-	x
A. hal	x	-	-	-	-	-	-	-	-	-	-
A. sic	-	x	x	-	-	-	x	x	-	-	-

*number of areas investigated

r-strategist (genus ranking is equal to 4 along a gradient from 1 to 4) (Ruf 1998). Species belonging to this genus are nematode-feeding specialists and are regarded as typical for soil in arable fields (Didden *et al.* 1994, Ruf 1998). The species *A. halleri* was highly dominating in arable field investigated by Brussaard *et al.* (1990), whereas *A. siculus* accounted for 25–35% of all predatory Gamasina in arable fields studied by Wasyluk (1975) and Paoletti (1988). It was frequently found also in green areas in Warsaw (Niedbała *et al.* 1990). Another nematode-feeder, *Eviphis ostrinus* (C.L. Koch 1836), presumed to be a forest species, was found in grasslands in two traditional farms (Słomian *et al.* 2005), as well as it occurred in *Tilio-Carpinetum* and a in meadow investigated by Niedbała *et al.* (1990), and in five of seven tree belts investigated by Seniczak *et al.* (1991a, b). However, it was not found by Paoletti (1988) in four residual woodlands and only single individuals have been recorded in two arable fields. This species has been found also in older stages of succession (forest stage) of post-industrial dumps (Madej 2004).

The genus *Paragamasus* comprises common, predatory, hemiedaphic species which prey on other mites, Collembola, and also small worms. The genus was found to be one of the dominant Gamasina in pasture, meadows and in wooded areas in two above-mentioned farms in SW Norway (Słomian *et al.* 2005, Gulvik and Madej unpublished). Five species of this genus have been reported from a natural wetland reserve (Gwiazdowicz and Klemm 2004) and seven species from a flooded forest reserve (Gwiazdowicz and Kmita 2004). Out of 10 *Paragamasus* species mentioned in at least one of the publications listed in Table 1, only one species of this genus was recorded by Niedbała *et al.* (1981) in forests, meadow and arable field. Only two species have been reported by Słomian (2002) in a deciduous forest reserve, whereas eight species have been recorded in older stages of primary succession in post-industrial dumps (Madej 2004). *Paragamasus misellus*, listed in five papers (Table 1), is not recorded from Norway (Gwiazdowicz and Gulvik 2005a).

Ruf (1998) considered Veigaiidae to be a ‘moderately’ *K*-selected family (ranking equal to 2 in Ruf’s system). *Veigaiia nemorensis* (C.L. Koch 1839), the species considered as typical for arable soils (Koehler 1999), was found to be highly dominating in a cluster of eight forests and one arable field (Ruf and Beck 2005). This species was the most numerous Gamasina in close-to-natural *Tilio-Carpinetum* and *Arrhenatheretum* meadow (0.3 and 0.5% of all Acari), and it has been found as single individuals in semi-natural *Tilio-Carpinetum* and in an old park; yet, it was not recorded in most of urban habitats and arable field (Niedbała *et al.* 1981, 1990). In Norway, this species has been recorded only in small numbers in pasture and meadow sites (Słomian *et al.* 2005, Gulvik and Madej unpublished).

Ruf and Beck (2005) noticed that generally the predatory mite communities found in arable fields are not reliably predictable due to the very small number of species and individuals. Large-scale studies by Madej (2004) have documented that during primary succession on post-industrial wastelands, the occurrence of Gamasina species is determined by random factors and little dependent on the of species occurring in adjacent habitats. However, all species that do occur in the youngest succession communities represent *r*-selected organisms (Madej 2004). Thus, the predatory mites can be used for large-scale assessment of ecosystem quality, but only when the database is sufficiently broad and they could be included into the other soil mites (e.g. Oribatida).

It can be concluded that species richness of Gamasina alone does not differentiate the types of land use. However, the percentage share of total Gamasina and the percentage contribution of species with a given life-history strategy are potential parameters that would be applied in monitoring systems. It would be a better tool to develop a reference data base than use of species richness separately.

3.6. Number of Gamasina species

It may be questionable whether the results obtained from small-scale studies are applicable at larger scales, especially if the

results are based on different number of samples from different areas and derive from heterogeneous habitats (Niedbała 2000). Rarely more than 60 species are recorded at a single site (Ruf and Beck 2005). Cancela da Fonseca (1991) documented that the number of species presented in each patch or site (local system, α -diversity) is in general lower (approximately by 50%) than their number in the ecosystem type (global scale, γ -diversity). This claim is certainly true for many results. For example, Buryn and Hartmann (1992) investigated three patches in a hedgerow and two in a surrounding meadow. They found a total 72 Gamasina species (65 species in the hedge and 41 species in the meadow). There were 50, 38 and 44 species in the patches of the hedgerow, whereas the patches of the meadow were inhabited by 26 and 34 species (α -diversity). In soil samples from four patches of restored wooded meadow (SW Norway), 35 species were recorded (20, 27, 27 and 29 in each patch besides 45 individuals not determined to species level) (Gulvik and Madej unpublished). Two habitats on post-industrial dump contained 38 species in total, (28 and 25 in each area) (Madej and Skubała 2002).

Generally, the species number of Gamasina is much lower in arable fields than in forests (Niedbała *et al.* 1981, Koehler 1997), but number of species varies between habitats. Twenty-eight Gamasina species were recorded in a beech forest on a very old (established in Middle Ages) post-industrial dump (Madej and Skubała 2002), whereas only 23 species occurred in a natural forest (*Tilio-Carpinetum*) (Niedbała *et al.* 1990). In four residual woodlands (*Quercus-Carpinetum*), only about 20 species were found in total (some specimens were not determined to species level) (Paoletti 1988). In a two-phase study (comparison between the first and the second samplings for forest, meadow and arable field), the community parameters such as abundance and species number are poorly reproducible (Ruf and Beck 2005). Thus, repetitions of standardized samples and appropriate statistical methods are essential.

3.7. Uropodina mites as bioindicators

Uropodina are seen as good indicators for the status of an ecological system (Koehler 1997, Błoszyk 1999, Madej 2004) and are associated primarily with habitats rich in organic matter and relatively mature ecosystems. The uropods are fungivores according to Werner and Dindal (1990), but can also prey (mainly on Nematoda) (Koehler 1997). Data regarding the diversity of Uropodina species from ecological quantitative studies are seldom presented in published papers. In four ecosystems investigated by Gulvik *et al.* (in press) in SW Norway, Uropodina were represented by four widespread European species. The density values of total Uropodina per square metre of soil showed no significant differences between sites, and the abundances of only two species (*Trachytes pauperior* Berlese 1914 and *Dinychus perforatus* Kramer 1886) differed significantly between the habitats. The percentage similarity values (Renkonen index) of Uropodina species were sufficiently different (ranging from 23 to 75%) and clearly separated the types of land use. It is possible that the high differentiation of the percentage share of Uropodina species is caused by the variation of their life-history tactics defined by Siepel (1994, 1995) (Gulvik *et al.* in press). For example, the thelytokous species *Trachytes aegrota* C.L. Koch 1841 (tactic X according to Siepel 1994) had the highest density at the traditional, old meadow (with constant but predictable disturbance), but did not tolerate recently changes in land use (like re-introduction of mowing and pollarding on abandoned wooded meadow).

In conclusion, it should be evaluated whether the Renkonen index has a high indicative value for measuring the distance between ecosystems containing an apparently similar composition of Uropodina.

3.8. Distribution of Macropylina mites

Nothroidea are the conservative oribatids of Macropylina group; they have low fecundity, long life spans, relatively stable populations and low dispersal rates (*K*-selected species) (Behan-Pelletier 1999). Maraun and Scheu (2000) have compared

the composition of Oribatida communities in 20 habitats which form a spectrum ranging from arable fields to different forest types. The dominance (percentage abundance) of Nothroidea (or Desmonata) was clearly greater in deciduous forests than in open habitats (pasture, fields and 4 yrs old fallow, see Figure 1 in Maraun and Scheu 2000). In three semi-natural birch forests, mean densities of *Nothrus silvestris* (Nicolet 1855) ranged from 100 to 3400 ind. m⁻² between different areas (Huhta and Niemi 2003). As yet, a single reason explaining the differences between forest sites investigated by Maraun and Scheu (2000) and Huhta and Niemi (2003) does not exist; more factors than humus type, age and forest type seem to affect densities of Nothroidea.

Relatively long period is needed to restore the species richness and density of Nothroidea communities after disturbance (natural or human). Huhta and Niemi (2003) studied birch stand planted after cultivation of arable fields and planted after clear-cutting of spruce stands and semi-natural birch forests. In the three 26–43 yrs old birch stands planted on cultivated soils, the mean density of *N. silvestris* was at least 10 times lower than in other birch stands. Gulvik *et al.* (2003) studied Nothridae and Camisiidae in semi-natural ecosystems in an agricultural landscape along a 100 km stretch of road in the extensively managed landscape (in inner part of Sogn region) in SW Norway. The samples were collected from abandoned traditional meadows and pastures, heavy overgrown traditional agroecosystems (about 50 yrs old deciduous forests with old pollards), and from neighbouring refugial habitats (areas which were probably inhabited by the native soil fauna when the surrounding area was under culture). In 65 of 232 samples, 11 species belonging to Nothridae and Camisiidae was found. The families mentioned above were found in 16–30% of samples from abandoned meadows and overgrown traditional agricultural habitats. Sixty percent of samples from refugial habitats contained these families. The number of species was relatively high: 6 species were recorded in samples from meadows and pasture (59 samples studied in total) and 8 species in samples

from refuges (in total 39 samples studied). Nine species were recorded in 134 samples studied from overgrown agricultural areas.

It seems that species richness of Nothroidea in the extensive managed landscape in inner Sogn (SW Norway) is greater than in deciduous forest sites and in human-made ecosystems studied in other European countries. To illustrate this assumption, lists of Nothridae and Camisiidae given in some papers and from author's own studies are showed in Table 2. Data presented in the Table 2 indicates that the more 'stressed' ecosystem in human-dominated landscape, the lower is the number of Nothroidea species (and percent of total abundance) in mite communities in the soil. In 13 habitats along the stress gradient (Table 2) studied by Niedbała *et al.* (1990), only four Nothroidea species were recorded. Nothroidea were not recorded in tree belt, arable field and the transitional zone between these two areas (Niedbała 1971), as well as in arable field (Wasylik 1986), and in *Arrhenatheretum* meadow in suburban zone (Żyromska-Rudzka 1976). These findings should stimulate further tests of the hypothesis that Nothroidea are good bioindicators for measuring stress-intensity across the landscape and in particular habitats (β - and γ -diversity).

Ptyctimina (ptyctimous mites), another oribatid group of Macropylina, cannot be used in the same way, because they are difficult to identify at the species level (Niedbała 2000), and literature does not present usually the lists of these species. Among ten Ptyctimina recorded in study made by Niedbała *et al.* (1990), nine species occurred in two *Tilio-Carpinetum* forests, whereas only one species was recorded in a meadow or in urban habitat. Paoletti (1988) reported ten species in four *Quercus-Carpinetum* woodlands and only one species in two out of nine maize monocultures. There were no ptyctimous species recorded in tree belt, arable field and the transitional zone between these two areas (Niedbała 1971), arable field (Wasylik 1986, Sgardelis and Usher 1994) and meadow (Żyromska-Rudzka 1976). From among nine ptyctimous species recorded by Gulvik *et al.* (in press) in four

Table 2. Patterns of Oribatida in different habitats (based on data from literature and author's own investigations). Abbreviations: Orib: Oribatida adults; Noth: Nothroidea; %: Percent of Nothroidea in total abundance of Oribatida adults; Not anal.: Not analysed; ?: Densities m^{-2} not given.

Habitats	Orib m^{-2}	% Noth of Orib	No of Nothri- dae and Camisi- idae species	Reference
8 patches at 3 meadows	3241m	2.5m	2 (0-1)*	Bielska (1986)
Old, traditional hay meadow	44 900	0.5	2	Gulvik <i>et al.</i> (2007)
50-y old deciduous forests (over- grown traditional, wooded hay meadow)	11 600	8.6	4	
20-y old abandoned traditional, wooded hay meadow	11 100	0.3	2	
Recently restored traditional, wooded hay meadow	6 400	0.2	1	
Total			4	
50-y old deciduous forests (over- grown traditional agricultural areas) with refuges	Not anal.	Not anal.	9	Gulvik <i>et al.</i> (2003)
Meadows	Not anal.	Not anal.	2	
Pastures	Not anal.	Not anal.	5	
Total			11	
Pasture	?	0.4	1	Hubert (2000)
Ditch near a field road	?	-	-	
7 patches in meadow (suburban zone)	?	0.5m	4 (0-3)*	Hubert <i>et al.</i> (2004)
Natural forest (<i>Tilio-Carpinetum</i>)	62 900	1.6	3	Niedbała <i>et al.</i> (1990)
<i>Tilio-Carpinetum</i> in suburban zone	14 900	-	-	
<i>Arrhenatheretum</i> hay meadow in suburban zone	24 900	-	-	
2 old parks, Warsaw	13 800m	0.2m	2 (0-2)*	
8 urban green areas, Warsaw	3 400m	0.1m	1	
Total			4	
4 residual woodlands	?	3.0m	2 (0-2)*	Paoletti (1988)
9 maize monocultures	?	0.7m	2 (0-2)*	
Total			3	
7 tree belts adjacent to arable fields	25 100m	Not anal.	4 (0-3)*	Seniczak <i>et al.</i> (1991a, b)
4 patches in tree belt	?	Not anal.	2	Sgardelis and Usher (1994)
Transitional zone	?	Not anal.	1	
2 transects in arable field	?	-	-	
Total	?		2	

^m mean for all areas

* number of species per area

habitats of traditional farm (from old traditional meadow to overgrown abandoned wooded meadow with old pollards), seven species were found in each habitat. It seems that Ptyctimina may serve as a bioindicator for quality of native, undisturbed soils and for evaluation of soil development in arable fields and in fallow.

On the example of Nothroidea and Ptyctimina, it can be stated that both number of species and percent of whole group decrease following human impact on the landscape. Although it may not be possible to determine the individuals to species level (like Ptyctimina), it is nevertheless possible to determine the abundance of taxa (e.g. as a percentage) in addition to frequency (the percentage number of samples containing the unit) in relation to the stress gradient in the landscape (input levels). Dominance and constancy of these groups in addition to species richness of Nothroidea may possibly be used as bioindicators for monitoring of long-term changes in soil at the landscape level. To characterize the species richness in particular habitat type (γ -diversity), it is necessary to conduct studies on the occurrence of Nothroidea in various patches of particular habitat type at the local/regional scale. Studies on changes in diversity of Nothroidea communities along a series of stress gradient and habitat types should be conducted.

3.9. Other Oribatida species

Many oribatids are eurybiotic species common (both in terms of abundance and frequency) in various habitat types ranging from forests to fields (low β -diversity), for example *Liebstadia similis* (Poronota). Therefore, it is not easy to find indicator species for monitoring the soil quality in relation to land degradation. However, some species have markedly different life-history strategy. Among these is thelytokously reproducing *Tectocepheus velatus* (X life-history tactic according to Siepel 1995) characterised by a short life-cycle, high reproduction rates and high colonisation ability (r -strategist). This species is often recorded as 'superdominant' in very heavily disturbed habitats (Behan-Pelletier 1999), but have also been abundant in forests (Siepel 1996). Numerous

populations of *T. velatus* (or *Tectocepheus* sp.) have been found in arable fields studied by Lagerlöf and Andrén (1985), Maraun and Scheu (2000), and by Kováč *et al.* (2001). High dominance of this species has been also recorded in a pasture under a heavy grazing regime (Clapperton *et al.* 2002), as well as in the youngest primary succession stages on a glacier foreland and post-industrial dumps (Skubała 2004, Skubała and Gulvik 2005). The numerous populations of this genus (which can account for up to 17% of all mites) occurred in urban habitats (Niedbała *et al.* 1990). *T. velatus* accounted also for 30–70% of the total microarthropod abundances in arable fields and shelterbelt (Olejniczak 2004). In a pasture in suburban zone, this species accounted for 38% of Oribatida adults (Hubert 2000) and made 27% of Oribatida adults in a meadow investigated by Żyromska-Rudzka (1976). Acidification of coniferous forest resulted in increase of the abundance of this species (Hågvar and Amundsen 1981). A very low percentage of *T. velatus* in old, traditional meadows (as low as $\leq 0.5\%$ of Oribatida adults) in SW Norway (Seniczak *et al.* 2007, Gulvik unpublished.) may suggest that semi-natural traditional meadows are ecosystems which are relatively little disturbed when compared to some heavily stressed ecosystems in the human-dominated landscape in Central Europe. Unfortunately, there are no reference data sets regarding occurrence of *T. velatus* along an input gradient in the rural landscape of Norway.

It can be concluded that *T. velatus* and other oribatids with a similar life-history tactic should be evaluated as potential bioindicators for measuring stress gradient in ecosystems.

3.10. Contribution of selected Oribatida taxa

The percent contribution of particular groups, like Poronota, *Tectocepheus* and Opiidae combined, could be the appropriate indicator of human impact. This procedure can be supported by applying the life-history tactics proposed by Siepel (1994, 1995, 1996) to evaluate the stress levels in ecosys-

tems. The Poronota include sexually reproducing *L. similis*, *Minuthozetes semirufus* (C. L. Koch 1841) and *Achipteria coleopterata* (Linnaeus 1758) (tactic XI, Siepel 1995). These species occur commonly in all terrestrial ecosystems, but are especially numerous in grasslands. Maraun and Scheu (2000) compared patterns of Oribatida communities in 20 habitats. Oppiidae, *Tectocepheus* and Poronota together made up at least 80% of all Oribatida in poor soil communities found in arable fields, pasture and fallow. High percentages of Poronota together with a pioneer taxa belonging to Gymnionota (such as Oppiidae dominated usually by *Oppiella nova* and/or *Tectocepheus* genus) would possibly characterise degraded ecosystems along an input gradient. Under new conditions, species with Spiel's tactic X (such as *O. nova* and *T. velatus*) increase in numbers faster than those with tactic IX, like the long-living Notothoidea species (Siepel 1995) which belong to the Macropylina. Thus they overcome the disturbance more easily than species with tactic IX (Siepel 1995).

The percent distribution of given Oribatida groups with different life-history strategy should be evaluated as a potential bio-indicator for measuring human effect on ecosystems. Application of the system of life-history tactics of mite species proposed by Siepel (1994) seems promising for monitoring ecosystem quality over time and across landscape.

3.11. Relative abundance of Oribatida and Actinedida

Generally, Oribatida respond in an opposite fashion to Actinedida when confronted with disturbances. Whereas densities and/or percentage abundances of Oribatida decrease, densities and/or percentages of Actinedida increase (Werner and Dindal 1990). The shift of dominance pattern from oribatids to actinedids has been also observed during the transformation of woodland into arable field in a temperate environment by Sgardelis and Usher (1994). Crossley *et al.* (1992) investigated microarthropod communities in areas undergoing secondary succession ('old field' in the early wood stage)

and arable fields established on this fallow. Before cultivation, the actinedids accounted for less than 1% of the mite community, but eight years later, the amount of Actinedida had increased up to 40% in both non-tillage and conventional agriculture. Battigelli *et al.* (2004) studied the impact of forest soil compaction and removal of organic matter on soil mesofauna communities. The relative abundance of Actinedida increased with treatment severity (compacted soil, removed organic matter), while the percentage of Oribatida decreased.

A similar trend has been recorded for four habitats under different land use regime in the traditional farm in SW Norway (Gulvik *et al.* 2007); the percentage of Oribatida was lowest (52%) and the percentage of Actinedida was highest (17%) in a recently restored wooded meadow (re-introduction of mowing and pollarding was the disturbance factor), whereas the opposite situation occurred in an old traditional meadow with predictable disturbance made by constant human activities (Oribatida 82% and Actinedida 7%). This pattern is compatible with the fact that most Oribatida are *K*-selected, while most Actinedida are *r*-selected species (Evans *et al.* 1961, Krantz 1978, Norton 1990, Siepel 1994, 1995, 1996). This is because the Actinedida have more generations during year than most Oribatida (Luxton 1981).

The ratios of absolute abundance of these two taxa can be compared across ecosystems. According to Werner and Dindal (1990), the values of Oribatida-to-Actinedida ratio falls below 1.0 in arable fields, and above 1.0 in more stable ecosystems (for example semi-natural grasslands or forests). Results from several studies support this claim. For example, the average values of ratio were 0.34 in patches of arable field and 4.27 in patches of adjacent tree belt investigated by Niedbała (1971). Eupodidae, Tarsonemidae and Tydeidae made the most numerous populations among actinedids found usually in arable fields (Lagerlöf and Andrén 1985, Werner and Dindal 1990). The most numerous populations of Tarsonemidae occurred in urban habitats and intensively managed meadow (Niedbała *et al.* 1990),

and in heavily grazed pasture (Leetham and Milchunas 1985, Kinnear and Tongway 2004). Luxton (1981) states that Tarsonemidae are ubiquitous, they do not prefer specific habitats or climatic factors, and they are able to make extremely numerous populations (4985 m^{-2} , 24% of all Actinedida). Tydeidae dominated Actinedida in grazed grassland (Clapperton *et al.* (2002). The Oribatida/Actinedida ratios were 11.8 in an old meadow in use, but only 3.1 in a recently disturbed habitat (restored traditional meadow on 20 yrs old fallow) in Western Norway (Gulvik *et al.* in press).

Percent contribution within microarthropods of Acari and Oribatida (dominance structure) of community is often used to illustrate differences between mite communities across ecosystems or treatments (e.g. Niedbala *et al.* 1990, Bengtsson 1994, Maraun and Scheu 2000, Battigelli *et al.* 2004). Ratio of absolute abundance of two taxa are likely an undervalued index (Krebs 1999). The ratios can be applicable where a full data for the whole community do not exist. The ratios of absolute abundance of Oribatid- to-Actinedida (or Oribatida to all mites and/or microarthropods) may be used for monitoring of long-term changes across the landscape ('early warning system'). Increases in actinedids may reflect recent disturbance in ecosystems (for example changes of land use). Perhaps, relative abundances (percentages and ratios) of actinedids and oribatids could be applied in monitoring of unique, species rich ecosystems and in monitoring system at large scale.

3.12. Measuring the biological quality of soil

Gardi *et al.* (2002) proposed the *Biological Index of Soil Quality* (called the QBS index). as a new approach to evaluate the soils. This index is based on the 'biological forms' of soil microarthropods. Each 'biological form' found in soil samples receives a score from 1 to 20 (so called *Eco-Morphological Index*, EMI), according to its adaptation to the soil environment. The QBS index sums up these scores. According to

Parisi *et al.* (2003), the QBS index is based on the following relation – the higher is soil quality – the higher will be the numbers of deep soil-living (i.e. euedaphic) microarthropod groups. The euedaphic microarthropods show morphological characters, such as reduction or loss of pigmentation and visual apparatus, streamlined body form, reduced and more compact appendages (hairs, antennae, legs), reduction or loss of flying, thinner cuticle etc. As a general rule, euedaphic forms get an EMI=20 whereas epiedaphic (surface-living) forms score an EMI=1. Authors state that the QBS index is a soil quality indicator which evaluates microarthropod groups present in soil samples and does not require a species-level diagnosis. The EMI-system gives Acari a top score, equal to 20, because it is almost impossible to obtain a soil sample free of Acari, and owing to the difficulties to outline easy-to-detect eco-morphological characteristics. Symphyla (order of the Arthropoda) also gets a top score of EMI = 20 (Gardi *et al.* 2002).

It is common knowledge that soil biota can increase or reduce agricultural productivity depending on their composition and as the effect of their different activities. For example, Symphyla feed, amongst other things, on plant roots and can become a major crop pest if their populations are not controlled by other organisms (Peachey *et al.* 2002, Edwards 1990). Symphyla is seldom mentioned in publications (Buryn and Hartmann 1992), probably because it is not a typical element of mesofauna and occurs, as a rule, sparsely in samples extracted from soils using the Tullgren-Berlese method (Edwards 1990). Gulvik *et al.* (in press) found up to 180 ind. m^{-2} of Symphyla, that was in contrast to $15\,500\text{--}83\,900 \text{ ind. m}^{-2}$ of Acari in the same areas. However, the QBS index does not calculate either quantity or quality of mite communities in soils. In light of these facts and due to the extreme species richness of mites and their great differences regarding life strategies, feeding guilds and niche differences as illustrated above, EMI values (20 for both Symphyla and Acari) seem difficult to accept.

4. MITES AS BIOINDICATORS IN A NATIONAL MONITORING SYSTEM – GENERAL PROBLEMS AND NORWEGIAN PERSPECTIVE

Studies on mites are very time-consuming; as many as 41 500 individuals of mites (14 000 Oribatida adults) have been recorded in 100 cores (5 cm in diameter, 5 cm depth) from a old meadow in SW Norway (Skubała and Gulvik unpublished). In addition to taxonomic problems discussed above, it is difficult to produce repeatedly data on mite communities from different habitat types on a national scale. Nevertheless, the large and conspicuous taxa as Nothroidea, Ptyctimina and Uropodina can be easily picked out and counted. Therefore, at least the frequency of these taxa in data sets from particular ecosystems, in addition to the recording the occurrence of some species, can be integrated into monitoring systems in national/international programmes. Most species belonging to these taxa are widespread in Europe (hol-arctic distribution), some are cosmopolitan and prefer low pH (e.g. *Platynothrus peltifer*), most of them occur commonly in forests, but inhabit open habitats too. On a regional/local scale, more laborious indicators need to be considered for monitoring the stress gradient across ecosystems, the valuable species, rich natural sites, or for identify ecosystems which should be preserved.

It is well known and documented that certain habitats are important for preserving the species diversity. Therefore, one can monitor the area and condition of selected habitats rather than monitoring the species directly. Biological parameters of soil can be suitable as tool to this goal. For example, a correlation between indices mentioned above (e.g. biodiversity of Nothroidea) and occurrence of Red List species should be examined in order to find soil bioindicators able to identify unique ecosystems or close-to-pristine ecosystems. Moreover, wood-associated mite species, e.g. species identified as rare in litter but occur numerous in woody debris from a given site [for example gamasid *Sejus togatus* (C.L. Koch 1839); Gulvik unpublished, Gwiazdowicz pers. commun.], should be incorporated into the monitoring programs. Research is

needed however to measure the weight of each of these indices in each of the habitat type according to stress-gradient across the landscape and in relation to landscape features (e.g. fragmentation).

Hedgerows found in many European landscapes (in some cases originating from Ancient Roman time, Paoletti 1988) and trees planted in rows, shelter belts, as well as patches of bush and herbs, have been and are the constant components of the traditional farming landscape in many European countries. These structures serve many purposes, including the improvement of biodiversity and provision of important refuges for native soil fauna. However, recently many agricultural landscapes are dominated by arable crops and early succession fields (fallow), and support only a few non-crop areas such as forest remnants and old grassland. It has been estimated that only 0.2% of Central Europe's deciduous forests remains in a relatively natural state (Bengtsson *et al.* 2000). In the Netherlands, it is no problem to find examples of high-input agricultural systems, but low-input agriculture is not so easily found, while virgin forests are absent (Siepel 1996).

In Norway, small, high-input areas (high-input meadows and arable fields) occur close to semi-natural vegetation with native (non-disturbed) soil in farmland managed as low-input systems. Unfortunately, there are no pristine deciduous forests that may serve as reference ecosystems in the surrounding landscape. Over large part of the country, traditional agrosystem production was stopped in the 1950–1960's, leaving the area to be recolonized by relatively young deciduous forests (now 40–50 yrs old). These forests cover today much of these old traditionally managed habitats. We can expect that species richness of mites has not been reduced to the same degree as in a landscape with a higher proportion of cropped areas. The natural soil refuges were always present, due to steep and stony terrain with clusters of trees and bushes, and the presence of old pollards in traditional farmlands. For example, Austad and Skogen (1990) reported that traditional, wooded hay meadows were established on areas dominated by large stones and brown soil with rich organic top layer between the

stones. The landscape complexity may compensate for loss of biodiversity caused by local management.

There are about 1200–1500 mite species recorded in Norway (Skartveit *et al.* 2002). Unfortunately, many records of species are inadequate because of unreliable identification. In addition, details about locality, vegetation type and microhabitat are absent in many older records, (like published in 1930's or before). Except the investigation of coniferous forest by Hågvar and Amundsen (1981) and Phytoseiidae (for review see Gwiazdowicz and Gulvik 2005a), Acari have been rarely the subject of studies in Norway. In some preliminary studies, 60 Gamasida species and 11 Parasitengona species new to Norway were found in relatively few samples (Małol and Gulvik 2002, Gwiazdowicz and Gulvik 2005b, Slomian *et al.* 2005). In contrast to many other countries, studies on soil mites have been sporadically used in applied soil ecology in Norway. Only the influence of pollution and manipulation of environments (application of liming and acid rain in coniferous forest) on mite fauna has been studied by Hågvar and Amundsen (1981), Hågvar and Kjøndal (1981), Hågvar and Abrahamsen (1990). There is a lack of published data on mite communities in deciduous forests and agrosystems.

The mite fauna in Norway is not expected to be as rich as the fauna in Central and Southern Europe. This is partly because of harsh climate and short growing season (northern latitude), and partly because the developing of ecosystems started about 8000 years ago and thus are relatively young. Nevertheless, in a relatively small number of samples taken as a preliminary survey, very rare species (*Saprosecans baloghi* Karg, 1969) can be found (Gwiazdowicz and Gulvik 2005b). In Sognefjellet mountain region (SW Norway) a very small population of very rare uropod species, *Iphidinychus gaieri* (Schweizer 1961) has been found (Gulvik and Błoszyk unpublished). Very few individuals of this species have been recorded only in the Alps (Schweizer 1961) and the Carpathians (Błoszyk and Olszanowski 1984).

Due to low human impact, SW Norway represents a region of high quality landscape due to the presence of highly diverse vegetation and semi-natural ecosystems under extensive management. In preliminary study on Gamasina in pasture and meadow in Sogn region, as many as 42 Gamasida species in 19 quantitative samples from tree hollows in pollards and 34 species in 11 quantitative soil samples (cores 22 cm², depth 5 cm) have been recorded (Slomian *et al.* 2005). Comparison of biodiversity in habitats across the rural landscape in Central Europe and in Norway might contribute to the development of indicator values which can be used to assess the landscape quality.

It can be concluded that research on mite fauna in the traditional agricultural landscape of Norway may provide new knowledge which may contribute to the development of monitoring systems for terrestrial ecosystems and the soil biodiversity in European landscapes.

5. CONCLUSIONS

1. Many studies have documented that composition of mite communities reflect the gradient of stress in the ecosystems and indicate the status of both biodiversity and natural resources. In practice, due to taxonomical difficulties and time-consuming operations, it is not possible to map and integrate into large monitoring systems the total biodiversity of mites at a national level. Nevertheless, some indices and selected taxa should be evaluated as potential bioindicators.

2. Most oribatid mites with their long life span, low fecundity, slow development and low dispersion can be the robust indicators of the state of the environment. Change in the dominance structure of mite communities or Oribatida-to-Actinedida ratio is suggested as an 'early warning' criterion for stressed mite communities.

3. Both number of species and the percentages of the whole Nothroidea and Ptyctimina groups decrease following human activity in the landscape. Even if the individuals cannot be determined to species levels (like Ptyctimina), it is possible to evaluate the dominance and frequency of taxa in relation to the human impact gradient (input levels)

within the landscape. *Tectocephus velatus* and other oribatids with a similar life strategy should be considered as potential bioindicators for impoverished ecosystems.

4. It seems that the percentage of Gamasida species with a given life strategy, as well as the Renkonen index (cf. similarity of Uropodina), is a promising tool for monitoring of changes in ecosystems across the landscape.

5. National and local reference datasets on the biodiversity of mite communities in diverse habitat types and along stress gradients have to be collected.

6. Residual natural biotopes (like woodlands, hedges, semi-natural grasslands) with species rich mite communities, found in rural and urban landscapes, should be preserved as refuges for dispersion of soil fauna.

7. Comparison of mite communities in traditional, low-input farmland in Norway and in the human-dominated landscape in Central Europe can contribute to a better understanding of how human activity has altered biodiversity along a land-use gradient, and will be of help in developing bioindicator values used in a soil monitoring system.

8. There is a need to develop standardised procedures for collecting of samples and analyses of datasets specifically adapted to ecological soil acarology.

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